INVESTING IN ECOLOGICAL INFRASTRUCTURE
METHODOLOGY OF VALUING CHANGES IN ECOSYSTEM CONDITION

Deliverable #4: Economic Value - Methodology Report

December 2015

Submitted to the Water Research Commission
by

Centre for Water Resources Research
University of KwaZulu-Natal

Project K5/2354
## Abbreviations

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
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<tbody>
<tr>
<td>CBA</td>
<td>Cost Benefit Analysis</td>
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<tr>
<td>COI</td>
<td>Cost of Illness</td>
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<td>CVM</td>
<td>Contingent Valuation Method</td>
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<tr>
<td>DALY</td>
<td>Disability-Adjusted Life Year</td>
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<tr>
<td>DWAF</td>
<td>Department of Water Affairs</td>
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<td>DWS</td>
<td>Department of Water and Sanitation</td>
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<tr>
<td>EPWP</td>
<td>Expanded Public Works Programme</td>
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<tr>
<td>KNP</td>
<td>Kruger National Park</td>
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<tr>
<td>KZN</td>
<td>KwaZulu-Natal</td>
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<td>NWRS</td>
<td>National Water Resource Strategy</td>
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<tr>
<td>NRM</td>
<td>Natural Resource Management</td>
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<td>RDM</td>
<td>Resource Directed Measures</td>
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<td>RDP</td>
<td>Reconstruction and Development Programme</td>
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<tr>
<td>SANBI</td>
<td>South African National Biodiversity Institute</td>
</tr>
<tr>
<td>TCM</td>
<td>Travel Cost Method</td>
</tr>
<tr>
<td>TEEB</td>
<td>The Economics of Ecosystems and Biodiversity</td>
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<tr>
<td>TEV</td>
<td>Total Economic Value</td>
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<tr>
<td>TWTP</td>
<td>Total Willingness to Pay</td>
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<td>URV</td>
<td>Unit Reference Value</td>
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<tr>
<td>VMP</td>
<td>Value of Marginal Product</td>
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<td>WRC</td>
<td>Water Research Commission</td>
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<tr>
<td>WTP</td>
<td>Willingness to Pay</td>
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<tr>
<td>WTA</td>
<td>Willingness to Accept</td>
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<tr>
<td>ZAR</td>
<td>South African Rand</td>
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Key Messages

1) **Investments in natural capital and ecological infrastructure are an integral part of the transition to, and the foundation of, a green economy**
   An understanding of the value of nature and how to include this value in public and private decision making is considered an important element of the transition to a green economy (ten Brink et al., 2012).

2) **For such investments to be successful, decision-makers need instruments and information to support decision-making and avoid inappropriate or unintended trade-offs**
   Demonstrating the contribution of ecological infrastructure and the value (or loss) of ecosystem services through case study examples can contribute to a more comprehensive and transparent evidence base of trade-offs in decisions. Such an evidence base can lead to better, more cost-efficient decisions, avoid inappropriate trade-offs and refocus economic and financial incentives to align with sustainability goals (ten Brink et al., 2012).

3) **Economic valuation is about valuing the change in the ecosystem as a result of the investment or the difference between the ‘with investment’ and ‘without investment’ alternatives**
   It is the value of the change in the ecosystem and ecosystem services as a result of the investment that is of interest, rather than an estimated value of the ecosystem itself.
   “Economic valuation is concerned with translating the physical changes in the ecosystem and the resulting change in ecosystem services into a common metric of associated changes in the welfare (utility or “happiness”) of members of the relevant population” (Heal et al., 2005:42).
   The valuation assessment should be framed in terms of changes in ecosystem services and benefits related to different investment options. The change may be related to a gain in ecosystem services and benefits with the investment, or a prevention of further loss of ecosystem services and benefits with investment. The baseline or benchmark for measuring the change is an important consideration.

4) **Economic valuation is context specific**
   Ecosystem benefit values are context specific and depend on the human preferences, institutional arrangements and the cultural setting in which the valuation takes place (Barbier et al., 2009). The context of the valuation and the intended use or purpose of the resulting values are key to ascertaining the type of valuation information required and therefore, the appropriate method to be applied. To be meaningful, the valuation must be a response to a specific question in a given context and should have a purpose and an intended audience. Important considerations include:
   - Scope – services/benefits and value types to be included in the study;
   - Spatial scale – delineation of the geographic extent and relevant population;
   - Temporal scale – the period of time over which benefits and costs are distributed.

5) **Both ecological and economic data are fundamental to the valuation process**
   Economic models should be used with biophysical models of ecosystem and service delivery changes. Ecological (biophysical) data is needed in determining the change in supply of services and benefits (quantity and quality changes); economic data is required in determining the willingness to pay for (or accept) the changes in services and benefits (Pendleton and Baldera, 2010). This implies the need for a multidisciplinary team working together to ensure that the ecological and economic models are aligned.
6) A variety of economic valuation approaches and methods exist
Economic valuation methods attempt to elicit human preferences for changes in ecosystems. The intended purpose of the valuation information, the ecological-economic relationships underpinning the relevant service(s) and benefits, and how the method influences the final estimate are key considerations in selecting the appropriate valuation method (Barbier, 2007). Ecological economics advocates an approach to valuation that integrates the perspective and methods of many disciplines (Costanza et al., 1997).

7) The distribution of the benefits and costs of the investment action should be considered
The assessment should identify who gains and who loses across affected stakeholder groups. Ethically, the analysis of the distribution of costs and benefits is important to ensure that interventions do not harm vulnerable people or lead to social exclusion (Pagiola et al. 2004). Practically, the costs and benefits received by local users can have a strong influence on how the ecosystem is used and managed and therefore the success and longevity of the action/investment (Pagiola et al. 2004).

8) Valuation under uncertainty and non-linearity related to ecological thresholds present both a challenge and opportunity for ecosystem valuation research
There is considerable uncertainty associated with both the functioning of ecosystems and the valuation of their attributes, as such identifying and dealing with uncertainty is an important consideration in ecosystem valuation. In the case of a change in the ecosystem near or across an ecological threshold, any (even small) change in ecosystem condition can lead to an abrupt and substantial change in the state of the system and a significant disruption in the provision of ecosystem services (Heal et al., 2005; Farley, 2012). Under such conditions, economic valuation may not be appropriate or may need to be addressed in a particular way.
1 Introduction

Ecosystems contribute long-term benefits to society and human wellbeing. In Deliverable 1 of this research project, the study team’s understanding of the relationship between natural capital and human well-being was illustrated in a diagram, reproduced in Figure 1, linking natural capital, ecological infrastructure, the green economy and human well-being.

![Relationship between natural capital, ecological infrastructure, the green economy and human wellbeing](image)

**Figure 1: Relationship between natural capital, ecological infrastructure, the green economy and human wellbeing.**

Note: The approach is that Natural Capital forms the resource base and that investments in ecological infrastructure allows the opportunities provided by ecosystem services and biodiversity to leveraged to support the Green Economy and Human Wellbeing. We recognize that in some models and perspectives, Biodiversity could be seen as an addition building block to natural capital, rather than something arising from it.

Source: Reproduced from UKZN (2014:2).

The United Nations Environment Programme defines a green economy as one that “results in improved human well-being and social equity, while significantly reducing environmental risks and ecological scarcities” (UNEP, 2011). A similar definition is adopted by the Water Research Commission’s Green Village - A Partnership Programme (WRC lighthouse 4), who highlight that a green economy is “modelled on acknowledging that ecological infrastructure and services have an economic value” (WRC, 2015). The transition to a green economy is supported by the current South African Government (UNEP, 2013) and viewed as “a sustainable development path based on addressing the interdependence between economic growth, social protection and natural ecosystem” (DEA, 2010:4). The green economy definition adopted by the South African Department of Environmental affairs is “a system of economic activities related to the production, distribution and consumption of goods and services that result in improved human well-being over the long term, while not exposing future generations to significant environmental risks or ecological scarcities” (DEA, 2010:4). A green economy “implies the decoupling of resource use and environmental impacts from economic growth” and “is characterized by substantially increased investment in green sectors, supported by enabling policy reforms. (DEA, 2010:4).
Ecological infrastructure refers to functioning ecosystems that generate and deliver ecosystem services (SANBI, 2014). The concept of ecosystem services has evolved as a means of relating ecosystem processes to human benefits (Brouwer et al., 2013). Ecological infrastructure is described as the equivalent of built infrastructure and represents the asset, or stock, from which a range of services flow (SANBI, 2014). Just as with built infrastructure, ecological infrastructure has to be managed and maintained to secure the flow of services, which requires investment in the form of time, effort and finance (SANBI, 2014). Given that such resources are limited, decisions and choices must be made on the allocation of resources with regards to ecological infrastructure.

In a TEEB (The Economics of Ecosystems and Biodiversity) report describing the transition to a green economy six ‘building blocks of a green economy’ are proposed, “Investing in environmental infrastructure” and “Proactive investment in natural capital (via restoration, conservation, and improved management practices)” are particularly relevant to this research project (ten Brink et al. 2012:vii, 34). The report advocates that "Investments in nature today can save money and promote economic growth in the long term and must therefore be seen as an integral part of the transition to and the foundation of a green economy." (ten Brink et al. 2012:vii). The report discusses several examples in support of the following conclusions:

- "Many activities can be more efficiently provided by maintaining or restoring ‘ecological infrastructure’ than by artificial structures or processes;
- Investing in natural solutions can also offer cheaper and more effective ways to mitigate the impacts of natural disasters than technical or man-made analogue solutions,
- Working with nature can also often offer cost effective solutions to address different policy goals and challenges (such as reducing carbon emissions” (ten Brink et al., 2012:26-27).

Working with nature is viewed as the centre of the transition to a green economy and the authors highlight that the “values of nature to economies and society have often been overlooked and not reflected in the decisions of policy makers, businesses, communities or citizens, contributing to the loss of biodiversity and subsequent impacts on people and the economy” (ten Brink et al., 2012:iii). All sectors of the economy are regarded as benefitting directly or indirectly from nature and should be engaged in the transition to a green economy (ten Brink et al., 2012). An understanding of the value of nature and how to include this value in public and private decision making is considered an important element of the transition to a green economy (ten Brink et al., 2012).

In justifying investment in and ecological infrastructure, there is a growing need to identify and value the benefits to society of investing in ecosystems (Weber and Stewart, 2009; Aronson et al., 2010; Pendleton and Baldera, 2010; Blignaut et al., 2013a; Verdone, 2015), and define the conditions and institutional arrangements under which the greatest benefits can be achieved (Goldstein et al., 2008; Suding, 2011). While the benefits of investing in ecosystems are frequently cited, their quantification and valuation is less common (van Zyl et al., 2004; Rey-Benayas et al., 2009; Blignaut et al., 2013a; Wortley et al., 2013). For example, in the context of ecological infrastructure, Wortley et al. (2013) reported that only 3.9% of ecosystem restoration evaluations considered the economic outcomes of restoration. Similarly, Aronson et al. (2010) found socioeconomic attributes of restoration to be underrepresented in the literature. Figueroa (2007) concluded that the economics of ecosystem restoration is not well studied. Several authors have expressed a need for research into the economic aspects of ecosystem restoration (de Groot et al., 2013; Wortley et al., 2013; Blignaut et al., 2014).

1.1 Report Outline

This report forms Deliverable 4 of the WRC project K5/2354 titled:
“Demonstration of how healthy ecological infrastructure can be utilized to secure water for the benefit of society and the green economy through a programmatic research approach based on selected landscapes”.

This report provides a review of (1) the literature of ecosystem valuation in South Africa and (2) the principles and theory of the economic valuation of ecosystems. The aim of the report is to suggest an approach and methods appropriate to investigating the economic aspects of investing in the ecological infrastructure of the uMgeni Catchment. Water-related benefits form the focus of the research. The project provides an opportunity to address an identified gap in knowledge around the economic aspects of ecosystem preservation, restoration and management, specifically within the context of the emerging concept of ecological infrastructure, and its importance in supporting the green economy.

The report is organised as follows. A review of the South African literature on ecosystem valuation follows this introduction. The third section is a discussion of the economic theory of ecosystem valuation, including an introduction to the methods of ecosystem valuation. The section closes with a discussion of how ecosystem valuation information can be used. In the fourth section, an approach to ecosystem valuation is proposed from a review of several valuation frameworks and guidelines. Challenges of ecosystem valuation are briefly introduced; valuation under the conditions of risk and ecological thresholds are discussed in greater detail. In the final section the study approach and case study options are described.

Several concepts and terms used regularly in this report require some clarification. ‘Investing in ecological infrastructure’ refers to actions taken to improve the condition of ecological infrastructure, maintain/preserve the existing condition and/or halt further declines in condition and may include physical interventions such as ecosystem restoration, rehabilitation, conservation and management activities as well as investments related to developing/strengthening institutions, influencing social behaviour amongst others.

The terms ‘restoration’ and ‘rehabilitation’, while theoretically different, share a similar focus (SER, 2004) and the terms are, at times, used interchangeably in the literature (Grenfell et al., 2007; Blignaut et al., 2010). In this report, where the discussion is a review of a study the use of terms follows the original study. Where the report refers more broadly to an action taken to alter or maintain the condition of an ecosystem, such as ecosystem management, protection, rehabilitation, restoration, reclamation and creation activities amongst others, without specific reference to either the starting or end condition of the system, the term ‘intervention’ is used.

Welfare and well-being are another two terms that are often used interchangeably in the literature. Welfare has two distinct meanings; one refers to aid or support for those in need or disadvantaged members of society. The second relates to health, happiness and good fortune¹, and is the meaning often used synonymously with well-being. In the Millennium Ecosystem Assessment (MA, 2005), ‘well-being’ is used; whereas in the TEEB (2010) report, ‘welfare’ and ‘well-being’ are used interchangeably. ‘Welfare’ is used in economics to describe human well-being and is considered quantifiable in terms of each individual’s own assessment of their well-being (Bockstael et al., 2000) and is related to utilitarian ethical theory (Perman et al., 2003). As such, the term ‘welfare’ is often associated with the economic interpretation of well-being and the economic methods and assumptions of measuring human well-being, where-as there are non-economic perspectives of what constitutes well-being and how well-being should (or should not) be quantified. For example, Potschin et al. (2014:20) define human well-being as “a state that is intrinsically and not just instrumentally valuable (or good) for a person or a societal group”. In this report, the term ‘welfare’ is used in the

context of the economic interpretation of human well-being, whereas ‘well-being’ is used as a more general term not restricted to the economic perspective.

2 Review of South African Studies

In South Africa, ecological rehabilitation activities are regularly undertaken, predominantly as part of the Natural Resource Management (NRM) programmes\(^2\) or as compliance with national environmental policies (Blignaut et al., 2013b). Under the Environmental Programmes of the South African Department of Environmental Affairs, 945,276 hectares of invasive alien plants were treated/cleared, 122 wetlands were under rehabilitation and 46,181 hectares of land were restored/rehabilitated during the 2013-2014 cycle (DEA, 2014). Total expenditure on the programme for the same period was approximately R3.1 billion. By 2012, the Working for Wetlands Programme (another of the NRM programmes), had been responsible for the rehabilitation of 906 wetlands across South Africa with a total investment of R530 million (Working for Wetlands, 2013).

Despite growing investment in the restoration of ecosystems in South Africa, evaluations of the outcomes of such activities are lacking (van Zyl et al., 2004; Ntshotsho et al., 2011; Cowden et al., 2013; McConnachie et al., 2013). Several South African studies have noted a paucity of information relating to the benefits of ecosystem restoration and identified the need to investigate the value of ecosystem restoration (van Zyl et al., 2004; Blignaut et al., 2013a). While there are several studies of the value of ecosystems in a given state, van Zyl et al. (2004:12) highlight that “studies specifically on the increased value associated with rehabilitation are unfortunately far less common”. The following section provides a discussion of studies related to the economic valuation of ecosystems in South Africa, with a specific focus on water-related ecosystems.

There is a distinction between economic evaluation and economic valuation. Economic evaluation involves ascertaining whether an investment is an efficient use of society’s resources and includes the identification, measurement, valuation and comparison of the costs and consequences of the alternatives being considered (Brouwer and Georgiou, 2012). Cost-benefit analysis is an example of an economic evaluation process. Economic valuation is the process of valuing the consequences of an investment and does not necessarily include the comparison of costs and benefits. As expressed by Young (1996:16) economic valuation “is the process of expressing preferences for beneficial effects or preferences against adverse effects of policy initiatives”. In terms of investing in ecological infrastructure, economic valuation is the process of determining the value of a change in an ecosystem, its components, or the services it provides as a result of the investment (EPA, 2009). One of the ways economic valuation information can be used is in the evaluation or appraisal of projects, interventions or policies (eftec, 2006).

In the South Africa literature, a number of studies have examined the benefits of alien vegetation clearing (or the costs of alien vegetation invasions) in South Africa, predominantly related to the national Working for Water programme (de Wit et al., 2001; Turpie et al., 2003; Marais & Wannenburgh, 2008; Currie et al., 2009; Gull, 2012). In a large national study Hosking (2010, 2011) investigated the value of recreational benefits associated with river water inflow to estuaries. The benefits and/or costs of river water quality changes have been considered (Turpie and Joubert, 2001; de Lange et al., 2012). While not linked directly to changes in ecosystems, the value of water in South Africa has received much attention (see for example Nieuwoudt et al., 2004 and Nieuwoudt and Backeberg, 2011). In addition, several manuals or guidelines related to the economic valuation and evaluation of ecosystems and ecosystem attributes have been developed for the South Africa context. The studies and manuals are discussed in the following section.

\(^2\) Part of the Government supported Expanded Public Works Programme (EPWP).
2.1 Alien Vegetation Removal Benefits

Gull (2012) undertook an economic assessment of specific changes associated with land rehabilitation in the Kromme River Catchment in the Eastern Cape, based on ecological modelling (Rebelo et al., 2012, 2015). The study considered three aspects:

- The costs and benefits of rehabilitation (Working for Wetlands and Working for Water activities);
- A comparison of proposed water supply augmentation schemes; and
- An investigation of the economic value of water to farmers in the context of possible water trading schemes.

The expected wetland rehabilitation benefits of reduced raw water treatment costs and increased life expectancy of a key water supply reservoir (Churchill Dam) were considered. It was found that the wetland rehabilitation benefits could not be valued as silt surveys of the dam were limited and a link between the change in wetland size and a change in water quality could not be established due to data limitations and a narrow time frame (Gull, 2012).

Additional land made available for agriculture and increased water yield were considered as expected benefits of the clearing of alien plants through the Working for Water activities. Improved reliability of water flow and reduced erosion were acknowledged as benefits, but were not valued due to insufficient data (Gull, 2012). The value of additional land was estimated through an analysis of the economic returns to land in the catchment. Gross margin analysis was used and sources of data included farmer interviews and local enterprise budgets. The study assumed that the future use of the cleared land would follow the existing land-use (proportion and type of land-use), but acknowledged that the assumption would likely result in an overestimation of the value as the agricultural potential of the cleared land had not been considered.

The value of the additional water yield was estimated by calculating the change in runoff due to the removal of alien invasive plants converted to a yield (cubic meter of water) per hectare cleared. The value of an additional cubic meter of water was taken as the cost to the relevant municipality of the 'next best' alternative water source option (i.e. the opportunity cost).

The costs and benefits of the Working for Water activities in the Kromme Catchment were compared through a cost-benefit analysis; the restoration was not economically viable over the 25 year period considered (Gull, 2012). Several points were noted: flow regulation and assurance of supply benefits were not valued; labour was a key project expenditure, but the associated socio-economic benefits (e.g. of indirect employment) for the local community were not included; and the risk and future costs of the spread of alien invasive plants (as a result of not clearing infestations) were not considered (Gull, 2012). Drawing on Nieuwoudt et al. (2004), the study emphasized the trade-off between yield and base-flow and that assurance of supply and reduced risk are important considerations in addition to yield (Gull, 2012).

Currie et al. (2009) investigated whether the water supply and assumed tourism benefits of restored fynbos habitat in the Assegaaibos mountain catchment of the Western Cape could justify the costs of clearing alien vegetation as well as restoring indigenous vegetation. Cost-benefit analyses were performed for three different restoration options. The water supply benefit was based on the water yield from the catchment before and after restoration and valued using the cost of bulk raw water. The value of the tourism benefit was estimated from the direct benefit obtained from permit sales for access to the Assegaaibos mountain catchment.

Fourie et al. (2013) focused on the value of an increased supply of marketable goods and services from alien plant clearing and restoration of indigenous (fynbos) vegetation in the Agulhus Plain,
Western Cape. Only marketable goods and services were valued and a market price approach was taken. A number of products are produced from the fynbos ecosystem of the Agulhas Plain and traded in both the formal and the informal sector, such as wildflowers, sour figs, honey-bush tea and honey. Net income at farm gate per hectare of each vegetation type was used to capture the direct benefits of wildflowers and other products originating from the fynbos ecosystem. The authors suggest that, since South African farmers are not supported by formal subsidies, net income at farm gate can be regarded as an efficient price. The change in net income was based on condensed hectares of cleared and restored land. Age until harvest for each vegetation type was included in the analysis.

The financial\textsuperscript{3} net present value to landowners of the clearing and restoration was estimated based on the additional direct income generated from harvesting fynbos flowers and other fynbos products and the direct costs of the programme. To estimate the economic net present value, the change in supply of sour figs (income from sour fig harvesting does not accrue to the landowner), impacts on the beekeeping industry and the opportunity cost of fuel wood were added into the model. While fynbos vegetation holds foraging value for bees, the services of the \textit{Eucalyptus spp.} forests (alien invasive trees) for beekeeping are greater, indicating an opportunity cost to the beekeeping industry if all \textit{Eucalyptus spp.} are cleared (Fourie et al., 2013). In addition, the removal of alien invasive trees incurs an opportunity cost in the form of forgone income to the community from harvesting fuel wood.

Based on the benefits and costs considered, both the financial and economic analyses yielded a negative return, indicating that the costs of restoration activities in addition to clearing alien vegetation could not be offset through the additional direct income of harvesting fynbos flowers and other fynbos products (Fourie et al., 2013). To further investigate the viability of alien plant clearing and indigenous vegetation restoration of the Agulhas Plain, the benefit of water released through the activities would need to be considered (Fourie et al., 2013). However, the value of the water supply benefit could not be determined as part of the study as it was unclear how water supply would change and information regarding the value of water across the various industries specific to the Agulhas Plain was not available. An alternative approach was adopted by estimating what the value of water would need to be to justify the alien vegetation clearing and restoration activities for various runoff scenarios (Fourie et al., 2013).

Tessendorf (2008) concentrated on the non-water related benefits of alien vegetation removal, specifically the biodiversity and ecosystem resilience benefits. Three Working for Water sites were selected for analysis, Port Elizabeth Driftsands (Eastern Cape), Worcester (Western Cape) and Underberg (KwaZulu-Natal). The contingent valuation method (CVM) was applied to value people’s preference for indigenous vegetation as a proxy of the value of increased biodiversity and ecosystem resilience. The estimated per hectare values (willingness to pay towards alien plant removal) differed across the study sites; where respondents were familiar with the Working for Water activities the willingness to pay bids were generally higher. The author noted that in some cases, respondents appeared to overstate their willingness to contribute out of a desire to encourage future employment opportunities. The study concluded that the CVM is an effective approach to estimating the economic value of the biodiversity benefit of the Working for Water alien vegetation removal activities, but that attention must be given to developing and applying the questionnaire. A comparison of the results to those generated using an alternative valuation technique was recommended as a means of testing the validity and reliability of the findings. The study drew on earlier work on the economic case for the Working for Water alien vegetation clearing programme by Hosking et al. (2002) on water related benefits and du Plessis (2003) on non-water related benefits.

A 2003 external evaluation of the Working for Water programme highlighted that monitoring and evaluation of the ecological outcomes of the programmes has been inadequate (Working for Water, \textsuperscript{3} See Section 3.1 for a description of financial analysis.)
In response, additional research has been conducted in relation to the ecological outcomes of the programme. A useful summary of, and links to, the research are provided on the Working for Water website\(^4\), including a study on the impacts of alien vegetation removal on the availability of water resources in the Western Cape and KwaZulu-Natal Provinces.

Law (2007) investigated the economic value to an urban population of the control of water hyacinth, an invasive aquatic plant, in the Nahoon River, East London. A benefit transfer approach was used to estimate the value of a reduction in water loss as a result of hyacinth control (drawing from van Wyk and van Wilgen, 2002). Recreational and non-use values were assessed through a contingent valuation study of two suburbs alongside the Nahoon River. The survey indicated to respondents that it would be possible to reduce the hyacinth invasion by 80%. The contingent valuation survey was applied in two phases (September 2006 and February 2007). Prior to the 2006 survey, heavy rainfall had washed much of the water hyacinth out of the system and there was an observable increase in the level of water hyacinth between the two phases (Law, 2007). As such, the study was able to examine the sensitivity of the willingness to pay (WTP) estimates to the change in water hyacinth level between the two phases. From the results, the study concluded that the increase in water hyacinth between the two phases did not have a significant effect on respondent WTP. Two interpretations of the result were offered, (1) most respondents had not noticed the increase in water hyacinth, indicating that non-use values drive the WTP for hyacinth control and (2) marginal increases in water hyacinth, far below an environmental threshold, result in very small marginal increases in WTP. An additional finding, similar to that of Tessendorf (2008), was that familiarity (or prior knowledge) of the problems linked to the plant invasion and reasons for removing or controlling infestations increased WTP towards control/removal activities. Law (2007:107) concluded that for the area studied, urban residents appeared to value the control of water hyacinth, and that the value is “predominantly based on the benefit that respondents acquire through the knowledge of a ‘healthy’ environment, or the non-use value of the resource”. The value information was used in a Cost-Benefit Analysis of the control of water hyacinth and the results indicated that the benefits outweighed the costs, even for the least cost effective control method.

### 2.2 Aquatic Ecosystem Rehabilitation Benefits

van Zyl et al. (2004) evaluated the impacts of urban river and wetland rehabilitation on property values and flood attenuation benefits across three case studies in Cape Town. While the hedonic pricing approach has often been applied, particularly in urban settings, to estimate the effect of environmental attributes on property prices, property sales data were inadequate for the statistical modelling of the hedonic pricing method (van Zyl et al., 2004). Instead, property price effects were assessed through structured estate agent interviews. Flood attenuation benefits were quantified for those properties affected by a shift in the 1 in 50 year flood line as a result of rehabilitation activities. ‘Damage costs avoided’ for developed properties and ‘preventative expenditure’ for undeveloped properties below the 1 in 50 year flood line prior to rehabilitation were used as a proxy for flood attenuation benefits. The potential costs of damage to structures and house contents from a 1 in 50 year flood were discounted over a 50 year period and multiplied by the number of developed properties to capture the damage costs avoided of raising the properties above the flood line. Preventative expenditure was estimated as the engineering costs of raising the undeveloped plots above the 1 in 50 year flood line.

The economic value of wetlands in South Africa has been investigated through several studies (Lannas and Turpie, 2009; Turpie and Malan, 2010). As an output of the Wetland Health and Importance research programme, Turpie and Malan (2010) provided a review of four wetland

valuation studies. Specific to wetland rehabilitation in South Africa, the WET-OutcomeEvaluate manual (Kotze and Ellery, 2008) describes the evaluation of six Working for Wetland rehabilitation projects located across South Africa. A cost-effectiveness approach was used to compare the costs of the rehabilitation to the outcomes achieved – specifically, the costs per hectare equivalent of intact wetland gained - as a result of the rehabilitation (Kotze et al., 2008).

In 2008, an assessment of the value of the livelihood benefits of the Manalana (Craigieburn) wetland (Pollard et al., 2008) was undertaken as part of the development of the WET-OutcomeEvaluate manual (Kotze and Ellery, 2008). The objective of the study was to provide an assessment of the livelihood benefits likely to accrue from the rehabilitation of the wetland (Pollard et al., 2008). The direct benefits to the community of the wetland from crop production, livestock grazing (safety net value), harvesting of wetland reeds, water for livestock consumption, and water for domestic use were investigated. The livelihood benefits of the wetland were quantified and valued, under the assumption that, if successful, the rehabilitation structures would secure the wetland upstream of the structures and safeguard the livelihood benefits. The value of the wetland rehabilitation was estimated as the difference between the value of the provisioning service and a scenario where rehabilitation was not undertaken and the wetland continued to degrade. The assumption was made that the rehabilitation of the wetland would maintain the supply of provisioning services, whereas continued degradation of the wetland would reduce the supply of services.

### 2.3 River Flows and Estuary Benefits

Hosking (2010, 2011) considered the allocation of river flows and investigated the value of recreational benefits associated with averting river water inflow reductions to estuaries. The contingent valuation method was used to assess 40 South African estuaries between 2000 and 2007. Estuary users were surveyed and asked to bid an amount of money (in ZAR) they would be willing to pay toward a hypothetical project to prevent a set of changes associated with a reduction in river inflow into the estuary. The likely future scenarios were developed using two sets of information:

- A prediction of changes in river water inflows to selected estuaries for the subsequent two decades (from DWAF, now DWS); and
- A forecast of how the changes would likely impact the recreational benefits of the estuaries (through a set of expert workshops).

A scenario was generated for each estuary and included aspects such as mouth closure (yes/no), changes in numbers of boaters, fishers, swimmers, bird watchers and loss of unique habitat. The additional water required per annum to avert the predicted reduction in river water inflow was ascertained. The travel cost method was applied as an alternative valuation method for comparison and credibility testing (Chege, 2009; Hosking, 2011).

As part of the broader study, the uMgeni Estuary was investigated by Chege (2009) who found a marginal willingness to pay for incremental freshwater inflow of R0.0015/m$^3$ (at 2007 prices). A decrease in inflows of approximately 23% from 2000 to 2025 was forecast for the uMgeni Estuary, leading to:

- Mouth closure for an extra 2-3 months per annum in winter;
- A 70% decrease is salinity during mouth closure and an 80% decrease in estuarine vegetation and loss of mangroves;
- An increase in water hyacinth growth during mouth closure;
- A potential increase in boating, but a possible 80% reduction of boating surface area due to water hyacinth encroachment;
- An 80% loss of angling fish, bait and birders (Chege, 2009).

It was estimated that 26.9 million m$^3$ of freshwater would be needed to avert the predicted scenario. The survey of estuary users was carried out in December 2007; 210 questionnaires were
administered, of which the results of 155 were considered valid. The results of the analysis are reproduced in Table 1.

**Table 1: Contingent valuation and travel cost estimates of the recreational value of the uMgeni Estuary, 2007 prices**

<table>
<thead>
<tr>
<th>Contingent valuation method results</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>User Population (households)</td>
<td>4414</td>
<td></td>
</tr>
<tr>
<td>Water required (million m³)</td>
<td>26.9</td>
<td></td>
</tr>
<tr>
<td>Mean WTP (Rand)</td>
<td>21</td>
<td></td>
</tr>
<tr>
<td>Median WTP (Rand)</td>
<td>9</td>
<td></td>
</tr>
<tr>
<td>Mean TWTP (Rand)</td>
<td>92 694</td>
<td></td>
</tr>
<tr>
<td>Median TWTP (Rand)</td>
<td>39 726</td>
<td></td>
</tr>
<tr>
<td>Mean value of water (Rand/m³)</td>
<td>0.0034</td>
<td></td>
</tr>
<tr>
<td>Median value of water (Rand/m³)</td>
<td>0.0015</td>
<td></td>
</tr>
<tr>
<td>Travel cost method results</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TCM of total WTP per (Rand/m³)</td>
<td>0.0010</td>
<td></td>
</tr>
</tbody>
</table>

Note: WTP – Willingness to pay; TWTP – Total willingness to pay; TCM – Travel cost method

To compare results across all the estuaries assessed, Hosking (2011) adjusted the estimated values to 2008 prices. The results – obtained from the contingent valuation estimates - are reproduced in Table 2 for selected studies. Hosking (2011) highlighted that the information generated through this type of valuation could be used to assess the efficiency of river-flow allocations into estuaries. As an illustration the author compared the value (median WTP estimates) of river inflow into the estuary to the value (median WTP) of abstracting river flow upstream of the estuary for the Keurbooms and uMngazi Estuaries. The comparison showed that while the marginal value of water flowing into the two estuaries was similar, the marginal value of upstream abstraction (or the opportunity cost) was different and higher for the relatively more developed region of the Keurbooms Estuary. Several qualifications were made:

- Efficiency is one consideration of what is socially desirable;
- Generalising across estuaries and time from findings specific to a given estuary at a given point in time should be undertaken with caution;
- Distributional aspects are an important consideration, in the case of the uMngazi Estuary significant differences in income exist between the upstream users, local people who are unable to pay the cost of potable water (the opportunity cost of inflows to the estuary) and visitors to the area, who are willing and able to pay to maintain inflows to the estuary (Hosking, 2011).

**Table 2: Predicted median willingness to pay for recreational benefits across selected South African estuaries, 2008 prices**

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Median predicted value of water (Rand/m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>uMngeni Estuary</td>
<td>0.0008</td>
</tr>
<tr>
<td>Average across all 40 estuaries</td>
<td>0.1130</td>
</tr>
<tr>
<td>Highest estimate (Kleinemonde West)</td>
<td>2.7676</td>
</tr>
<tr>
<td>Lowest estimate (St Lucia)</td>
<td>0.0000</td>
</tr>
<tr>
<td>Average across 37 estuaries (excl. outliers)</td>
<td>0.0340</td>
</tr>
</tbody>
</table>


A research programme was undertaken on the ecology and economics of the East Kleinemonde estuary, Eastern Cape, with the aim of ascertaining both the value of the estuary and the relationship between the value and the condition of the estuary. Direct use values of recreation and the non-use
(or existence) value of the estuary were investigated (Turpie et al., 2009). The direct use value of the estuary was estimated by examining actual expenditure by holiday makers, and through application of the hedonic pricing method to determine property value effects. Conjoint valuation was then used to estimate the potential change in value associated with a change in condition of the estuary based on four attributes; water level, water quality, numbers of angling fish and the amount of saltmarsh versus water weed. The existence value of the estuary was considered through a contingent valuation assessment of people’s willingness to pay (WTP) for the preservation of a key species associated with the estuary (the endangered Estuarine Pipefish). Two payment options were investigated: WTP an amount included in the monthly water bill over and above monthly water charges (payment to the municipality) and a once-off payment for a water supply option that would reduce water demands on the estuary. Significant differences in WTP estimates were found between the two payment options; respondents appeared less willing to pay the municipality for conservation, attributed to concerns over poor service delivery (Turpie et al., 2009). This finding draws attention to the potential influence of external factors on WTP estimates.

In a discussion of river flow management in the uMgeni system, Still et al. (2010) highlighted the value to the regional economy of several sporting events (canoe races) which depend upon flow releases from one of the dams on the uMgeni River. Based on the value to the regional economy of the races and the estimated marketing value to South Africa of the internationally recognised Dusi Canoe Marathon, Still et al. (2010) estimated the value per cubic metre of water released (above the base flow) required to secure these races - ZAR4.76 per cubic metre in (in 2008 rands). In comparing this value to the economic output per cubic metre of water used in other sectors, Still et al. (2010) found the value of water for the recreation events to be greater than that for general water use. However, the authors cautioned that the estimated value was not representative of the value that event participants would be willing or able to pay for the water. The study highlights the competing and complimentary nature of water uses (and water values).

In considering the contribution of ecological infrastructure to the economy, Crafford and Hassan (2014) developed and applied economic production functions to measure the relationship between elements of estuarine biodiversity and the recreational fishery economy in KwaZulu-Natal (KZN). The authors contend that economic valuation techniques for regulatory or intermediate services are not well developed, but that economic production functions are the most commonly used method to value the marginal contribution of intermediate services. The biological production of fish was specified as a function of the number of species present in the system and the estuary type. The number of species present in the system was modelled to vary with differences in key ecosystem component and process variables: estuarine condition, shoreline length (as a proxy for the extent of tidal flat area), estuarine type and nutrient cycling. The model parameters were estimated using existing data: time series data for fish egg abundance and mean annual rainfall, and cross-sectional data on the compositional and structural elements of KZN estuaries. The marginal contribution of estuarine components and services to fish catches were then derived from the estimated model parameters. The estimated marginal effects can then be used to calculate shadow values based on fish catch prices (Crafford and Hassan, 2014). The authors highlight that the resulting shadow values would be an overestimate of the marginal contribution of ecosystem services to fish catch as the economic inputs to harvesting the fish were not captured in the production function (due to insufficient data). Furthermore, seasonal effects were not included in the model, but are expected to explain a portion of the variation in fish production (Crafford and Hassan, 2014). Key conclusions drawn included: the usefulness of existing databases in providing evidence of the relationships between ecological infrastructure and the economy (no primary data collection was undertaken); the role of estuarine and marine ecosystems in supporting fishery production and how ecological degradation may indirectly

\[5\] Shadow price or value is the opportunity cost to society of participating in some form of economic activity. It is applied in circumstances where actual prices cannot be charged, or where prices do not reflect the true scarcity value of a good (DEFRA, 2011:65). Often, shadow prices are market prices adjusted to account for market distortions such as tariffs, quotas and monopolies (Mullins et al. 2014).
impact valuable industries; and the potential for using environmental variables as additional predictors of fish stocks. In a related study, Hassan and Crafford (2015) highlight the importance of long-term monitoring programs and systematic data collection and advocate for efforts to align the collection of economic and ecological data with regard to ecosystems.

2.4 Water Quality Benefits

de Lange et al. (2012) estimated the costs associated with water pollution in the Olifants River Water Management Area. The study focused on the implications of salinisation on commercial irrigated agriculture and microbial pollution on the general population of the area. A production function approach was taken to estimate the income lost when crops are irrigated with polluted water. The contribution to revenue of the last unit of irrigation water applied (the value of marginal product (VMP) of irrigation water) at different levels of salinity was estimated for four crops. The VMP was shown to decline with incremental increases in salinity for maize, potato and citrus, but not for wheat. Wheat has a relatively high salinity threshold, whereas citrus was shown to be the most sensitive to salinity. From the results, de Lange et al. (2012) indicated that, should the salinity of irrigation water increase in future, a shift in cultivation towards wheat production could be expected.

In the same study, a cost of illness (COI) approach was applied to estimate the impacts of microbial pollution, but it was noted that contingent valuation and averting behaviour methods have also been applied in valuing the impacts of waterborne diseases (de Lange et al., 2012). The COI approach “attempts to measure benefits of pollution prevention and/or reduction by estimating the direct and indirect opportunity costs associated with an illness” (de Lange et al., 2012:244 citing Pegram et al., 1998). The incidence of diarrhoea and the health-care costs per treatment were used to estimate the direct costs. A distinction in the cost per treatment was made between cases with access to above-RDP standard services and those subjected to below-RDP standard services (higher treatment cost). Indirect costs were taken as the decrease in productivity of human capital as a result on the burden of disease, estimated using the disability-adjusted life year (DALY) measure and taking gross income as a proxy for the value of human capital. From the results, the authors showed that the direct health costs of diarrhoea exceeded the costs of construction of a water treatment plant with sufficient capacity to supply potable water to the population of the study area with below-RDP standard water and sanitation services. It was concluded that investments to improve access to safe drinking water in the study area would realise net savings from a social welfare perspective. The study considers the value of preventing the existing microbial pollution and therefore, takes a total value approach; the value of a unit reduction in microbial pollution is not estimated.

Turpie and Joubert (2001) considered the tourism value of the Kruger National Park (KNP) and the relationship between river water quality and tourism value. Three components of tourism value were examined: on-site expenditure (i.e. within KNP), off-site expenditure (attributable to the resource under consideration) and consumer surplus. The three approaches differ mainly in terms of whose benefits they reflect (Turpie and Joubert, 2001). From the results the author’s indicated that approximately 30% of tourism value would be lost if the rivers became completely degraded. To determine the potential impact of a change in river quality (in contrast to complete degradation) on tourism value, a combined contingent - conjoint valuation approach was applied. The conjoint valuation method was used to examine how the tourism value of the KNP would change in response to changes in river quality represented as changes in the level of selected attributes. Recreational users were surveyed and asked to rate combinations of four river related attributes. The conjoint model can then be used to generate a utility score for any combination of the attributes (Turpie and

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6 The Olifants Water Management Area corresponds with the South African portion of the Olifants River catchment and includes portions of Gauteng, Mpumalanga and Limpopo provinces.
Joubert, 2001). Usually, a cost variable (based on the cost of a trip) is included in the conjoint model to translate the utility score to a monetary value (Turpie and Joubert, 2001). Due to the high variability in trip costs associated with KNP, a cost variable could not be included in the analysis. Instead, two contingent valuation-style questions were included in the survey and used to relate utility to expenditure by providing values for the ‘ideal’ and ‘worst’ scenarios relative to the status quo. The relationships between river attributes and utility (conjoint analysis) and between utility and expenditure (contingent valuation) could then be used to estimate the expected changes in trip expenditure for different combinations of river attributes. The authors highlighted that the relationship between utility and expenditure was derived from the regression of only three data points (the status quo, worst, and best scenarios); and recommended future research/studies would need to address this shortcoming. The contingent valuation model was based on visitors responding to a change in river related attributes by adjusting the duration of their stay; as a further recommendation the authors suggest that the possibility of a change in the frequency of visits should be considered.

An additional objective of the Turpie and Joubert (2001) study was to provide a demonstration of the economic valuation of environmental attributes. In this regard, they concluded that “it can be relatively straightforward to assess the current total value of a natural amenity… the main complexity lies in evaluating the potential impacts of a change in the resource” (Turpie and Joubert, 2001:397).

### 2.5 Valuing Water

Two reviews of research into the value of water in South Africa, Nieuwoudt et al. (2004) and Nieuwoudt and Backeberg (2011), provide a useful overview and discussion of the modelling approaches taken to determine the value of water across water use sectors of the country. The more recent review, Nieuwoudt and Backeberg (2011) found the main use sectors studied to be agriculture, forestry, municipalities (domestic consumption) and the environment. Environmental water use was related to the valuation of estuary services (Hosking, 2010, 2011). Methods used to model water use included operational research, econometric analysis, input/output analysis, willingness to pay and the conceptual framework of water markets. The review found water values to differ significantly between sectors, and between and within geographic areas; it was noted that care should be taken in comparing water values estimated using different tools.

### 2.6 Economic Assessment Manuals

Several manuals or guidelines related to the economic valuation and evaluation of ecosystems and ecosystem attributes have been developed for the South African context.

Turpie and Kleynhans (2010) developed a protocol for the quantification and valuation of wetland ecosystem services. Guidelines are provided for the valuation of the provision of natural resources; flow regulation; water quality amelioration; recreation and tourism; scientific and educational value; and cultural, spiritual and existence value. Three criteria are advocated for determining the required comprehensiveness of the study, the scope (coverage of different values), the extent (how beneficiaries are defined and value expressed) and methodological rigour (accuracy), all of which are informed by the purpose of the study (Turpie and Kleynhans, 2010). The manual generally takes a total wetland value or ‘with’ or ‘without wetland’ approach, in contrast to considering the value associated with a (unit) change in the level of service or benefit delivered (marginal value).

A framework and manual specific to evaluating the trade-offs in allocating water to various uses, including aquatic ecosystems, was developed by Ginsburg et al. (2010). The framework is aligned with the South Africa National Water Resource Strategy (NWRS) which follows from the National Water Act. A key element of the NWRS are the Resource Directed Measures (RDM) that define the
goals and objectives for the desired condition of water resources in aquatic ecosystems. Stemming from the RDMs is the concept of a Management Class which classifies the quality and overall health of the water resource under various levels of use. The Ginsburg et al. (2010) manual relates specifically to evaluating the trade-offs of a change in management class and the resulting change in delivery of aquatic ecosystem services.

Four phases of the evaluation process are outlined, systems analysis, assessment of ecological change, valuation of ecosystem services, and the evaluation of trade-offs, with a step-by-step guide to tasks within each phase. In the second phase of the process, a comparative risk assessment is undertaken (generally through an expert workshop setting) to establish the links between the ecological outcomes of different management scenarios and changes to the ecosystem services and benefits that the system delivers (Ginsburg et al., 2010:27). The comparative risk assessment is suggested as a systematic approach to “describing the effects of ecological change on human well-being that is transparent, clearly recorded and repeatable” (Ginsburg et al., 2010:58). The authors note (citing CIC, 2007) that the comparative risk assessment process has been used as an initial step in the economic valuation of environmental resources. The proposed evaluation process is demonstrated through an application to two case studies: a rapid cost-benefit analysis of the direct costs and benefits of a proposed dam on the Klein Dwars River, Greater Sekhukhune District Limpopo; and a comprehensive assessment of a hypothetical dam on the Sand River, Bushbuckridge Limpopo, to investigate the effect of the dam on the provision of ecosystem services. In an additional research report, Ginsburg et al. (2012) review and propose methods for extracting information from research to support decision making on social-ecological systems.

A manual for cost benefit analysis in South Africa with specific reference to water resource development has been designed, revised and updated by Mullins et al. (2014). The manual provides a set of guidelines for conducting cost benefit analysis (CBA) specific to evaluating the development and management of water resources in South Africa. The intended purpose of the manual is to provide practical guidelines to decision-makers in the public sector. Included in the manual is a discussion of methodologies to calculate the economic value of water for various water uses; applications and limitations of CBA with practical examples; and a list of shadow and surrogate prices for South Africa.

van Zyl and Leiman (2002) developed an economic evaluation framework specifically for evaluating water conservation/water demand management measures in South Africa. The framework was tested in a comparison of two water conservation/water demand measures, in terms of the cost per unit of water saved, and found to be robust.

The ecological, hydrological and economic effects of ecological restoration were assessed at eight existing restoration sites across South Africa (Blignaut et al., 2013b; summarised in Table 3). The findings were used to construct a framework and model for analysing decisions regarding the restoration of natural capital (Crookes, 2012; Crookes et al., 2013). The goal of the study was to provide a method of selecting and prioritising restoration activities.

While drawing on the standard benefit-cost evaluation framework as a starting point, system properties and risk were built into the model through three stages:

- a system dynamics model was developed and used to maximise the net present value of each of the eight case studies;
- risk analysis using Monte Carlo simulations were conducted on the model to determine the risk profile of each restoration project;
- The outputs from the system dynamics model and Monte Carlo simulations were plotted on a portfolio map (net present values were plotted against the probability of technical success of the project) as a visual representation of risk versus reward for each project.
Considering three risk profile maps (the probability of success, standard deviation and coefficient of variation), the study found that the projects with the highest potential payoffs were those projects where downstream water consumers benefit from the restoration activity (Crookes et al., 2013).

The authors suggest that, by incorporating risk, the proposed framework provides a more thorough means of evaluating restoration activities compared to the standard net present value approach, but note that only the economic viability of the project is considered (Crookes et al., 2013). However, as with conventional Cost-Benefit Analysis, the approach does not integrate alternative motivators of restoration, such as legal compliance and job creation, which would need to form part of the final investment decision (Crookes et al., 2013).

Table 3: Summary of ecological restoration studies at eight existing restoration sites across South Africa

<table>
<thead>
<tr>
<th>Study site</th>
<th>Economic activity</th>
<th>Disturbance</th>
<th>Valuation method(s)</th>
<th>Benefit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agulhas (Fourie et al., 2013)</td>
<td>Wild flowers, Apiculture</td>
<td>Alien plant spread</td>
<td>Market value</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Water yield</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Firewood</td>
</tr>
<tr>
<td>Beaufort West</td>
<td>Agriculture (Sheep, Goats)</td>
<td>Alien plant spread</td>
<td>Value of water – adjusted municipal tariff</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Water yield</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Bio-electricity</td>
</tr>
<tr>
<td>Drakensberg</td>
<td>Agriculture</td>
<td>Erosion</td>
<td>Value of grazing capacity – market-based</td>
<td>Agricultural production</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Agricultural services</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td>Water quality</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Soil carbon</td>
</tr>
<tr>
<td>Namaqualand</td>
<td>Mining</td>
<td>Destruction of natural vegetation</td>
<td>Report not available</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td>Sheep farming</td>
<td></td>
<td></td>
<td>Soil carbon</td>
</tr>
<tr>
<td>Kromme river (Gull, 2012)</td>
<td>Agriculture (Sheep, Cattle, Vegetables, Honeybush tea, Fruit)</td>
<td>Damage to wetlands Alien infestation</td>
<td>Economic returns to land - gross margin analysis Opportunity cost of an additional cubic meter of water (cost to the municipality for an alternative source)</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Water yield</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Water quality</td>
</tr>
<tr>
<td>Lephalale</td>
<td>Agriculture</td>
<td>Reduction in fire frequency, fencing that causes bush encroachment</td>
<td>Report not available</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Game production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Bio-electricity</td>
</tr>
<tr>
<td>Oudtshoorn</td>
<td>Ostriches</td>
<td>Destruction of natural vegetation</td>
<td>Market-based, farm-scale models</td>
<td>Agricultural production</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Soil carbon</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Ecolabelling</td>
</tr>
<tr>
<td>Sand River</td>
<td>Forestry, agriculture</td>
<td>Alien spread and exotic plantations</td>
<td>Report not available</td>
<td>Water yield</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Eco-tourism</td>
</tr>
</tbody>
</table>

Source: Adapted from Blignaut et al. (2013b).

Blignaut et al. (2013b) conclude that there are no universal decision models that can be applied to identify and prioritise ecological restoration projects. The authors recommend that decision-making should consider both the risks (economic, social and ecological) and returns of a proposed activity and address key elements of governance, social/beneficiary and environmental aspects.
In a recent study undertaken to identify the costs and benefits of restoring and managing ecological infrastructure in the uMgeni River catchment KwaZulu-Natal, the Unit Reference Value (URV) measure was applied as one approach. URVs were calculated for three discrete water services (water supply, baseflow and sedimentation avoidance); the authors suggest such values can be used to compare water services supply options such as new storage dams, waste water recycling works and desalination plants for example (Jewitt et al., 2015). In South Africa, the Unit Reference Value (URV) measure, developed in the 1980s as a method for planners to assess proposed water works schemes, is commonly applied to assess the feasibility of structural or non-structural water resource interventions (Niekerk and du Plessis, 2013). The measure has also been applied in evaluating the feasibility of water demand management and catchment management measures and in forecasting the relative costs of new water supplies into the medium- and long-term (Niekerk and du Plessis, 2013).

3 Economic Theory

3.1 Economic Analysis

Economic analyses related to decisions regarding ecosystems (such as whether to undertake ecological restoration) generally fall within two broad categories (1) economic benefit assessments and (2) economic impacts and contribution assessments (Bendor et al., 2015). The focus of economic benefit assessments is measuring or determining the economic value produced through the activity (market and non-market value), whereas economic impact and contribution studies examine the impact on the economy of expenditure related to the activity (gross output and employment) (Pendleton and Baldera, 2010; Bendor et al., 2015). Within an economic framework, increases in employment are seldom viewed as an economic benefit of a policy or project intervention, but are captured through an economic impact assessment (Pendleton and Baldera, 2010). In an economic evaluation of a proposed project or policy, information from both types of assessments may be considered, however it is important to note the distinction between the two and, importantly, that measures of economic impacts do not accurately reflect economic value (Pendleton and Baldera, 2010; Bendor et al., 2015).

An additional point to note when considering the value of ecosystem benefits, is that the value created through an investment can be determined from the perspective of society or from the perspective of a private enterprise (Gittinger, 1982). The benefits of ecosystems are likely to vary across different groups within society and there are likely to be different incentives regarding ecosystem management between the public as a whole and private groups. Economic analysis aims to establish the net economic returns (costs and benefits) to society and provides information on the net benefit to society as a whole of an activity or investment (Gittinger, 1982; Hitzhusen, 2007). Economic values reflect the worth of a good, service or investment to society (Emerton and Bos, 2004) and are measured in terms of opportunity cost (the value of a good or service in its next best alternative use) or in values determined by the willingness to pay (Gittinger, 1982). Financial analysis provides information on the (private) profitability of an investment and is used to determine the net returns to private equity capital (Gittinger, 1982; Hitzhusen, 2007). Financial values, also referred to as private values, reflect the worth of a good, service or investment to a particular enterprise (WBCSD, 2013:22). While the approach to comparing the costs and benefits of an investment is the same for both analyses, what is considered as a cost or a benefit differs. In financial analysis, taxes and interest on borrowed capital are treated as costs, and subsidies and interest earned on equity capital are considered benefits (Gittinger, 1982). In economic analysis, taxes and subsides are treated as transfer payments (Gittinger, 1982). Financial analysis is generally based on market prices, whereas economic analysis is based on adjusted market prices and non-market values (estimated values where no market prices exist). ‘Market price’ relates to an amount of money actually paid for a good or service, it is the portion
of value realised within the market place and does not necessarily capture scarcity value and therefore the real cost or benefit to society (Gittinger, 1982; Fisher et al., 2011; WBCSD, 2013; Mullins et al., 2014). To reflect the value to society, market prices are generally adjusted based on the opportunity cost to society (Gittinger, 1982).

In a decision-making context, the choice between analytical alternatives should not be a case of ‘either or’, but rather a combination of approaches best suited to the context of the decision. The appropriateness of the approach depends on the question being addressed and the intended purpose of the information (see section 3.5 for a discussion on the various uses of value information); no single type of information can be regarded as ‘best’ for all contexts (eftec, 2006). For example, the distributional effects of a decision or investment can be captured in an economic evaluation by applying a weighting system across different groups of society. However, distributional analysis is not the key focus of economic evaluation and distributional effects may be better captured through deliberative and participatory methods (eftec, 2006). Economic analysis should be viewed as one of the criteria available for evaluating a particular course of action (Bockstael et al., 2000).

### 3.2 Economic Value

In a traditional economic framework, the economic value of ecosystems is derived from the principles of neoclassical welfare economics. Debate over the strict assumptions and limitations of this approach has extended the ‘valuation of ecosystems’ beyond the traditional economic framework. Within the emerging field of ecosystem valuation, two perspectives are generally recognised, neoclassical economics and ecological economics (discussed further in Sections 3.2.1 and 3.2.2 respectively). In addition, there are several sub-disciplines of economics which focus on different elements. Those most common in the literature of ecosystem valuation are introduced in Table 4.

**Table 4: Sub-disciplines of economics particularly relevant to environmental aspects**

<table>
<thead>
<tr>
<th>Sub-discipline</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Environmental economics</td>
<td>Draws from neoclassical welfare economics and the study of market failure. Primarily concerned with identifying externalities and evaluating regulatory policies designed to control them. Cost–benefit analysis is regularly used to inform policy and decision-making, as a practical vehicle for applied welfare economics.</td>
</tr>
<tr>
<td>Natural resource economics</td>
<td>Draws from neoclassical growth economics. Primarily concerned with the efficient and optimal use of natural resources particularly as inputs into production. Applied in governing common-pool natural resources and finding dynamically optimal rates of renewable or non-renewable resource extraction.</td>
</tr>
<tr>
<td>Ecological economics</td>
<td>Recognizes that economic and environmental systems are interdependent. Studies the joint economy–environment system in the light of principles from the natural sciences. Addresses the social goals of sustainable scale, fair distribution, and efficient allocation.</td>
</tr>
<tr>
<td>Institutional economics</td>
<td>Focuses on understanding the role of institutions in shaping economic behaviour. Primarily concerned with the organization and control of the economy and its power structure (whose interests count).</td>
</tr>
</tbody>
</table>

Source: Drawn from Perman et al. (2003); Hackett (2006); The New Palgrave Dictionary of Economics (http://www.dictionaryofeconomics.com).

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7Ecological economics is not intended to be a sub-discipline of either economics or ecology, but rather a transdisciplinary approach integrating the thinking and methods of many disciplines (Costanza et al., 1997a).
The economic view of ecosystem value takes an anthropocentric utilitarianism perspective of value, whereby the value of an ecosystem is derived from the contribution it makes to human well-being (MA, 2003). Other theories of value hold, for example the intrinsic value theory, whereby an ecosystem is considered to possess value unrelated to its instrumental use or contribution to human well-being (Heal et al., 2005). Furthermore, even within the anthropocentric view of ecosystem value, there is debate on how ecosystems contribute to human well-being and how these values should (or should not) be expressed (Bockstael et al., 2000; de Groot et al., 2006; Gómez-Baggethun and Ruiz-Perez, 2011). Fisher et al. (2008) argue that an economic framing of ecosystems and ecosystem services is logical in that economics is the study of how humanity provides for itself and natural systems are fundamental to sustaining human life. Gómez-Baggethun & Ruiz-Perez (2011:614) describe economic valuation as a “pragmatic and transitory short-term tool to communicate the value of biodiversity using a language that reflects dominant political and economic views”.

There are many misconceptions about the term “economic valuation”, one being that economic valuation refers only to an assessment of the market or commercial value of an ecosystem service (Heal et al., 2005). The economic view of ecosystem value is conceptually much broader than is often recognized and can capture human-preferences for a range of benefits and includes components that have no market value (Heal et al., 2005). While economic valuation does not include all sources of value, theoretically it does encompass a broad range of values as set out in the Total Economic Value (TEV) framework (see Section 3.2.3) including non-use values (such as option, bequest and existence value). The Economics of Ecosystems and Biodiversity (TEEB, 2010) suggest that the economic perspective should be used as a complement to other forms of valuation and ethical and scientific arguments for conserving or restoring ecosystems (TEEB, 2010).

3.2.1 Neoclassical welfare economics

The economic valuation of ecosystems and the services and benefits they provide originates from neoclassical welfare economics. Within this framework, the objective is to maximise human welfare as measured by utility, a representation of an individual’s level of satisfaction. Two key assumptions are made:

• It is the individual who is the best judge, based on their own values and preferences, of whether an event increases or decreases their utility (Perman et al., 2003; Hackett, 2006); and

• Individuals take decisions to maximise their utility in the presence of constraints (Farber et al., 2002).

Determining the social benefits and costs of an intervention or policy change is thus contingent on measuring impacts in terms of the satisfaction of individual preferences (Hitzhusen, 2007). An assumption being that the economic welfare of society is based on the economic welfare of its individual citizens (Young, 1996). An individual’s preferences for a proposed change (intervention) can be quantified through his/her willingness to pay (WTP) for an improvement in welfare or willingness to accept compensation (WTA) for a reduction in welfare (Young, 1996). The basic assumption being that the individual would be willing to trade (to receive or to give up); or put another away, that the “the different sources of value that affect the individual’s utility are potentially substitutable” (Heal et al., 2005:48).

In neoclassical economics, the goal of economic efficiency is used in allocating scarce resources; an allocation of resources is regarded as economically efficient (or Pareto optimal) if it is not possible to make one additional person better off (improve welfare) without making at least one other person worse off through a reallocation of the resources (Young, 1996; Perman et al., 2003). A reallocation of resources is considered desirable if it increases at least one individual’s utility (welfare) without reducing another’s, a Pareto improvement (Perman et al., 2003). In practice, it is seldom the case that...
a reallocation of resources will not result in some individuals losing. Compensation tests have been developed as a pragmatic approach to evaluating alternative allocations of resources and are used in the application of welfare economics to environmental issues (Perman et al., 2003). The Kaldor–Hicks–Scitovsky test states that for a reallocation of resources to be desirable two conditions must hold: (1) the gainers must be able compensate the losers and still be better off; and (2) the losers must not be able to compensate the winners for forgoing the reallocation and be no worse off than they would have been if it did occur (Perman et al., 2003). Compensation tests do not require that actual compensation occurs, but that the potential exists. In the application of compensation tests, gainers and losers are treated equally, the fairness of the distribution of well-being is not considered (Perman et al., 2003).

Within a neoclassical economics framework, economic efficiency is satisfied in a perfectly functioning competitive economy (Young, 1996). In practice, many of the necessary conditions do not hold and actual market economies differ from the ideal circumstances in a variety of ways, known as ‘market failure’. Welfare economics is used to identify and recommend ways of correcting for ‘market failures’ so that economies perform better in relation to the goal of efficiency (Perman et al., 2003:124). Assigning economic value to ecosystems is a form of market correction aimed at improving allocative efficiency.

In summary, economic value, as viewed from the neoclassical welfare economics perspective, is the expression of individual preferences under conditions of scarcity (Farber et al. 2002) and is used in welfare economics “to assess the efficiency of a proposed change from the point of view of society’s welfare” (Brouwer and Georgiou, 2012:430).

### 3.2.2 Ecological economics

As described in Section 3.2.1, the focus of traditional neoclassical economics has been the goal of allocative efficiency. While generally accepting the neoclassical theory regarding efficiency, ecological economics contends that the neoclassical approach neglects the issue of scale (physical size and scale of the economy) and does not adequately address the goal of fair distribution (Costanza et al., 1997a).

Ecological economics recognizes three goals of society: efficiency, fair distribution, and sustainable scale (Costanza et al., 1997a; Costanza, 2000), where:

- Efficiency refers to the allocation of resources to alternative uses to maximise utility or human welfare, as determined through neoclassical economic theory;
- Fair distribution refers to the division of resources (as final goods and services) fairly, or in such a way as to limit the degree of inequality, and includes consideration of future generations and non-human species;
- Sustainable scale refers to maintaining the physical flow of matter-energy from low-entropy raw materials to high-entropy wastes at sustainable levels (ensuring the magnitude of human activities remains with the bounds of what is ecologically sustainable) (Daly, 1992; Costanza, 2000).

Ecological economics takes the view that the economic system is a part of (as opposed to distinct from) the global ecological system (Costanza et al., 1997a; Perman et al., 2003) and the expansion of the economic system is limited by the size of the finite global ecosystem (Munda, 1997). Economic and ecological systems are seen as interdependent and the sustainability of the interactions between the two systems is a core issue in ecological economics (Costanza et al., 1997b). The multiple values of ecosystems (ecological, economic, socio-cultural) are acknowledged: “there are multiple values which, in principle, may be equally correct and fundamental, and yet in conflict with each other” (Gómez-Baggethun et al., 2014:7). The idea of value pluralism recognizes that “valuation processes
in social-ecological systems involve dealing with multiple and often conflicting valuation languages, whereby values may be combined to inform decisions but may not be reduced to single metrics” (Gómez-Baggethun and Barton, 2013:238).

Ecological economics is not intended as a sub-discipline of either economics or ecology, but rather as an approach that integrates the thinking and methods of many disciplines (Costanza et al., 1997a). Methodological pluralism is advocated (Norgaard, 1989; Venkatachalam, 2007). Chan et al. (2012) for example, propose a framework for incorporating the cultural values of ecosystems into the valuation process, drawing on narrative methods, weighting of benefits approaches and discourse based valuation. In considering the valuation of urban ecosystem services, Gómez-Baggethun and Barton (2013) discuss ways of measuring and capturing the economic, socio-cultural, and insurance values of urban ecosystems and how these values can be used to inform urban planning. Castro et al. (2014) adopted a spatial analysis approach to compare ecosystem service trade-offs across biophysical (ecosystem service supply) and socio-cultural and economic (demand) dimensions.

Costanza et al. (1997a:87) describe ecological economics as “a new way of looking at the problem that can add value to the existing approaches and address some of the deficiencies of the disciplinary approach. It is not a question of ‘conventional economics’ versus ‘ecological economics’, it is rather conventional economics as one input (among many) to a broader transdisciplinary synthesis”. Munda (1997), Ma and Stern (2006) and Venkatachalam (2007) provide additional discussion on the differences between the ecological economics and environmental economics (based on neoclassical welfare economics) perspectives.

3.2.3 Categories of economic value

Neoclassical economics distinguishes between two broad sets of value, use and non-use values, which together represent total economic value (TEV) (Turner et al., 2003). Heal et al. (2005) contend that conceptually ‘total economic value’ encompasses many values, a characteristic not always recognized outside the economics profession. Further categories of value are defined within the TEV framework and brief descriptions of several are given in Table 5.

<table>
<thead>
<tr>
<th>Table 5: Categories of economic value</th>
</tr>
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<tbody>
<tr>
<td><strong>Category</strong></td>
</tr>
<tr>
<td>Direct use value</td>
</tr>
<tr>
<td>Indirect use value</td>
</tr>
<tr>
<td>Option value</td>
</tr>
<tr>
<td>Quasi-option value</td>
</tr>
<tr>
<td>Existence value</td>
</tr>
<tr>
<td>Altruistic value</td>
</tr>
<tr>
<td>Bequest value</td>
</tr>
</tbody>
</table>

Source: Drawn from Pearce and Warford (1993:100); Barbier et al. (1997:83); Perman (2003:402); and Turner et al. (2008:43).

These categories tend to vary across analysts and may over-lap, for example bequest value can be viewed as an option value (for potential future use of the service) or as a non-use value (that it exists.

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8 Norgaard (1989) describes and builds an argument for methodological pluralism.

9 ‘Total’ refers to the sum of its components, not the total value of the ecosystem (eftec, 2006).
for future generations regardless of its use value) (Turner et al., 2008), and is sometimes included under existence value (Barbier et al., 1997).

3.2.4 Measures of economic value

The way economic values are expressed will depend on the intended purpose of the valuation information. A key distinction is between marginal and total values; each provide different information to stakeholders (Nieuwoudt et al., 2004). The total value of an ecosystem refers to the value people derive from the ecosystem compared to not having it (Verdone, 2015). The use of a ‘total value’ approach is limited in the context of ecosystem valuation as it is challenging or even nonsensical to determine what it would mean to be ‘without’ an ecosystem (Pendleton and Baldera, 2010). Marginal value refers to the change in value resulting from an additional unit of a good or service produced or consumed (DEFRA, 2011; Brouwer et al., 2013). Important to note is that marginal value and average value, while being expressed as ‘value/unit’, are not synonymous. Average value is total value spread across all units of the resource or service; each unit has the same average value. Marginal value is the value of the last additional unit of resource or service added and depends on the scarcity of the service/resource.

For marginal analysis to hold true, the ‘next unit’ should not be capable of tipping the ecosystem over a functional threshold10 (Fisher et al., 2008). The scale of the ‘next unit’ must be meaningful (i.e. not the global population) and appropriate to the decision or research question (Fisher et al., 2008). Whether marginal or total/average values are estimated will depend on the economic model and method used (Nieuwoudt and Backeberg, 2011) and should be considered when selecting the appropriate valuation method.

From an economic perspective (efficiency goal), it is marginal value that provides the basis for selecting between alternative investments (Bockstael et al., 2000; Turner et al., 2003; Fisher et al., 2008; Bateman et al., 2011). With reference to ecological restoration, Verdone (2015:5) assert that “Restoration decision-making is not based on the Total Economic Value of a landscape, but rather on restoration’s ability to change that value. When identifying areas of restoration potential it is important to know how much the value of ecosystem goods and services would change if the landscape were restored”. For the contribution of ecosystem services to be readily integrated into policy and economic decision-making, marginal values are needed (Turner et al., 2003). Fisher et al. (2008) indicate that ecosystem service studies have largely focused on the current supply of services (total value) rather than applying marginal concepts and evaluating changes in ecosystem service delivery. The authors suggest that time series data on land cover could be a useful way of incorporating marginal change into ecosystem service assessments. An example of such an approach is the ecosystem service value study of land-use change on Chongming Island, China (Zhao et al., 2004). The study used satellite imagery analysis combined with a value transfer approach11 to investigate the value of changes in ecosystem services supply with changes in land-cover/land-use over time.

3.3 Ecosystem Services

The ecosystem services concept has emerged as a framework for articulating the relationship between ecosystems, their processes, and human well-being. Numerous definitions and typologies of ecosystem services have been proposed (Haines-Young and Potschin, 2009). A detailed discussion is provided by Fisher et al. (2009) who, drawing from Boyd and Banzhaf (2007), define ecosystem

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10 There is much debate around ecological thresholds and how to identify when a threshold may be reached (Turner et al., 1998; 2003), see Section 4.1.1 for a discussion.

11 See Section 3.4.4 for a description of the benefit/value transfer approach to ecosystem valuation.
services as “the aspects of ecosystems utilized (actively or passively) to produce human well-being” and highlight that:

- Ecosystem services are ecological phenomena including both ecosystem structure and ecosystem process/functions;
- An ecological phenomena only becomes a service once it is required or used (directly or indirectly) by at least one individual; and
- Ecosystem services are not necessarily benefits, benefits are generated from ecosystem services, but generally in combination with other forms of capital input.

Recreational activities such as angling, boating or swimming, for example, are benefits that arise from a combination of inputs, including the contribution of the ecosystem (fish, clean water) and manufactured and/or human capital (equipment or life guards) (Boyd, 2007). This distinction is important in considering that only a portion of the value of certain benefits may be attributable to the underlying ecosystem service or attribute (Mace et al., 2011).

A further distinction can be drawn between intermediate and final services and is proposed as a way of avoiding the double-counting of benefits (Boyd and Banzaf, 2007). An example of this distinction is provided by Tuner et al. (2008), Figure 2. Whether a particular service is an intermediate or final service will depend on the benefit being considered (Turner et al., 2008).

![Figure 2: An illustration of intermediate ecosystem services, final ecosystem services and ecosystem benefits](source: Reproduced from Turner et al. (2008:6).)

Haines-Young and Potschin (2010) proposed the ‘cascade model’ to demonstrate the relationship between ecosystem structure and human well-being. The model has since been adapted and used by others (Martin-lopez et al., 2014; La Notte et al., 2015), including in the approach of The Economics of Ecosystems and Biodiversity (TEEB) study (de Groot et al., 2010). The TEEB adaption places the ‘cascade model’ into a socio-cultural context and emphasises the role of institutions and human preferences in relating biophysical elements to human benefits. Vatn (2005) maintain that institutions influence the decisions people make at all levels of society. The model, Figure 3 illustrates the distinction between biophysical structures and processes and benefits to humans and the role of the ecosystem services concept in linking these two aspects. Primmer et al. (2015) suggest that the way in which ecosystem services are governed influences the relationship between ecosystems, ecosystem services and benefits. The authors argue that the element of governance is

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12 Institutions are the “rules and conventions of society that facilitate coordination among people regarding their behaviour” (Bromley, 1989:22 cited by Vatn, 2005:10). “They appear as conventions, norms and externally sanctioned rules” (Vatn, 2005:6).
missing from the 'cascade model' and propose an extension of the model to incorporate ecosystem service governance.

Figure 3: The ‘cascade model’ of the relationship between ecosystem structure and processes and human well-being

Source: Reproduced from de Groot et al. (2010), adapted from Haines-Young and Potschin (2010).

Not all ecosystem services are complimentary. The use of an ecosystem to provide a certain service (e.g. effluent treatment by a wetland) may preclude other ecosystem services/uses (for example using the same wetland for recreation) (Turner et al., 2003; Westerberg et al., 2010). A change in an ecosystem may increase the supply of some services while decreasing others, or may prevent a loss in service supply rather than increasing supply (i.e. an intervention may have maintained the supply of services by halting further degradation) (Pendleton and Baldera, 2010). A comparison of the ‘before and after’ situation may not reveal the value of the intervention; a ‘with or without’ comparison is necessary. There are also dis-services associated with ecosystems, and certain services may be viewed as beneficial by some individuals, but not by others.

The concept of ecosystem services implies that ecosystems are assets that produce a flow of services overtime, just as conventional economic and financial assets (Barbier, 2007). This perspective is aligned with the theory of ecological infrastructure as an asset from which a range of services flow. An understanding of ecosystem services is useful in conceptualizing how ecological infrastructure contributes to human well-being and in identifying how changes in ecological infrastructure may affect the benefits humans derive from ecosystems (eftec, 2006). Ecosystem benefit values are context specific and depend on the human preferences, institutional arrangements and the cultural setting in which the valuation takes place (Barbier et al., 2009:249).

3.4 Valuation Methods

Economic valuation methods attempt to elicit human preferences regarding changes in ecosystems and ecosystem services. Most commonly, these preferences are expressed using a monetary metric,
however, alternative metrics can be used (King and Mazzotta, 2000). For example, Fisher et al. (2008) suggest an index of vulnerability, happiness or number of lives saved could be used instead of monetary units. The intention is to express values in a common metric to facilitate comparison (eftec, 2006). Monetary units are widely recognized and readily understood by decision-makers and the general public. Using a monetary metric to express ecosystem service and benefit values does not suggest that only ‘money-generating’ services have value and are considered, but rather the purpose of ecosystem valuation is to measure and express benefits that are not reflected in the current market system (Pagiola et al. 2004).

A variety of economic valuation methods have been developed, some are applicable to specific issues or data sources, while others can be applied more broadly. The intended purpose of the valuation information, the ecological-economic relationships underpinning the relevant service(s) and benefits, and how the method influences the final estimate are key considerations in selecting the appropriate valuation method (Barbier, 2007). More than one method may be applied within a valuation assessment. Given the existing extensive base of literature on ecosystem valuation methods (Bateman et al., 2011), the purpose of this section is to introduce the basic methods, provide examples of how they have been applied and draw attention to their strengths and weaknesses. Barbier et al. (1997), King and Mazzotta (2000) and Turner et al. (2008) provide detailed reviews of ecosystem valuation methods; Heal et al. (2005) focus specifically on the application of valuation methods to aquatic ecosystem services and Young (1996) and Lange and Hassan (2006) review methods of valuing water.

Primary economic valuation involves the collection of data specific to the ecosystem(s), services and beneficiaries under consideration (Brouwer et al., 2013). Primary valuation methods may be market or non-market based (Turner et al., 2008). Secondary valuation involves the transfer of value information from primary valuation studies of similar systems or sites to the site under consideration (Brouwer et al., 2013) and is generally referred to as benefit or value transfer. Economic valuation methods are introduced in the following sections and a summary table is provided for each category of methods.

Where possible the individual methods have been related to an example from the South African literature discussed in Section 2 (see table heading ‘application in SA’).

### 3.4.1 Market-based valuation

Market-based valuation methods make use of existing market behaviour and market transactions to derive the value of ecosystem services and benefits (Turner et al., 2008). Two methods are noted, market price and production function methods (see Table 6 for an overview). The market price approach makes use of existing prices for goods and services traded in the market for example reeds harvested from wetlands and fynbos products. The standard economic theory of consumer and producer surplus is used to estimate the economic value of the ecosystem good or service using market price and quantity data (King and Mazzotta, 2000, Turner et al., 2008).

Where market prices are distorted (as a result of subsidies for example) prices may need to be adjusted to reflect the true economic value of the good or service (Barbier et al., 1997), sometimes referred to as the efficiency or shadow price method. The efficiency price method may also be used to address issues of equality by assigning distribution weights (Barbier et al., 1997).

The production function approach treats an ecosystem good or service as an ‘input’ into the production of a marketed ‘output’ (good or service) (Barbier, 2007). The value of the ecosystem service is derived from the change in the production of the marketed good or service as a result of changes in the provision of the ecosystem service. A production function is specified which relates the

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13 As yet, there is no single classification system for ecosystem valuation methods, various groupings and terminologies exist (de Groot et al., 2006); the categories used here may differ from other information sources.
production of the output to its inputs or, similarly, a cost function can be specified relating the costs of producing the output to the costs of the inputs (Turner et al., 2008). The approach is used to estimate economic values for non-market ecosystem services that contribute to the production of marketed goods or services (King and Mazzotta, 2000; Heal et al., 2005). For example, in the case of irrigated agricultural crops, the economic benefits of improved water quality can be derived from the increased revenues from greater agricultural productivity (King and Mazzotta, 2000; de Lange et al., 2012). Barbier (2007) provide a useful discussion of the production function approach through an application to value the effect of a change in coastal wetland habitat area on the market for commercially harvested fish in Thailand. South African applications are reported in de Lange et al. (2012), Crafford and Hassan (2014) and Hassan and Crafford (2015). Heal et al. (2005) highlight the use of the production function approach in valuing the benefits of coastal wetland-fishery linkages and provide a detailed review.

Table 6: An overview of market-based valuation methods

<table>
<thead>
<tr>
<th></th>
<th>Market price</th>
<th>Production function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Description</td>
<td>Existing market prices are used (or adjusted) to</td>
<td>Infers the value of ecosystem services from the change</td>
</tr>
<tr>
<td></td>
<td>value ecosystem services traded in the market</td>
<td>in the production of market goods as a result of changes</td>
</tr>
<tr>
<td></td>
<td></td>
<td>in the provision of the ecosystem service.</td>
</tr>
<tr>
<td>Suitable for</td>
<td>Cases where the ecosystem services (or good) is</td>
<td>Cases where the ecosystem service is an input in the</td>
</tr>
<tr>
<td></td>
<td>traded in the market and market data are available.</td>
<td>production of market goods or services; most suitable when</td>
</tr>
<tr>
<td></td>
<td></td>
<td>the ecosystem input contributes significantly to market output.</td>
</tr>
<tr>
<td>Value measure</td>
<td>Direct and indirect use values.</td>
<td>Variable depending on study.</td>
</tr>
<tr>
<td>Value category</td>
<td>Price, quantity and cost data (time series); data</td>
<td>Data on input use and output to specify the production</td>
</tr>
<tr>
<td></td>
<td>on factors that affect demand (e.g. income).</td>
<td>function and market structure; how the quantity or quality</td>
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<td></td>
<td></td>
<td>of the ecosystem input affects the costs of production</td>
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<td></td>
<td></td>
<td>for the final good, the supply and demand for the final</td>
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<tr>
<td></td>
<td></td>
<td>good and the supply and demand for other factors of</td>
</tr>
<tr>
<td></td>
<td></td>
<td>production.</td>
</tr>
<tr>
<td>Data requirements</td>
<td>Market data are often readily available; estimates</td>
<td>Data are often readily available; method is robust and</td>
</tr>
<tr>
<td></td>
<td>based on observed data of actual consumer</td>
<td>conceptually straightforward.</td>
</tr>
<tr>
<td></td>
<td>preferences; robust.</td>
<td></td>
</tr>
<tr>
<td>Strengths</td>
<td>Limited to cases where ecosystem services/goods</td>
<td>Limited to valuing those services that can be used as</td>
</tr>
<tr>
<td></td>
<td>are traded in the market; prices may need to be</td>
<td>inputs in the production of marketed goods. Information</td>
</tr>
<tr>
<td></td>
<td>adjusted to correct for distortions; cannot be</td>
<td>is needed on the relationships between improved quality or</td>
</tr>
<tr>
<td></td>
<td>easily used in the case of larger scale changes</td>
<td>quantity of the ecosystem service and the output - these relationships may not be well</td>
</tr>
<tr>
<td></td>
<td>likely to affect the supply or demand for a good</td>
<td>known or understood.</td>
</tr>
<tr>
<td></td>
<td>or service; may overstate benefits if the market</td>
<td></td>
</tr>
<tr>
<td></td>
<td>value of other inputs used to bring the ecosystem</td>
<td></td>
</tr>
<tr>
<td></td>
<td>service/good to the market are not deducted</td>
<td></td>
</tr>
<tr>
<td>Challenges</td>
<td>Generally used to value ecosystem ‘goods’ such as</td>
<td>Widely used to estimate the impact of ecosystem condition</td>
</tr>
<tr>
<td></td>
<td>forest products and fish, can be applied in the</td>
<td>on recreation and agriculture activities.</td>
</tr>
<tr>
<td></td>
<td>case of certain recreational benefits.</td>
<td></td>
</tr>
<tr>
<td>Example</td>
<td>Fourie et al. (2013) - value of an increased</td>
<td>de Lange et al. (2012) - implications of salinization of</td>
</tr>
<tr>
<td></td>
<td>supply of marketable goods and services from</td>
<td>irrigation water on commercial irrigated agriculture;</td>
</tr>
<tr>
<td></td>
<td>alien plant clearing and restoration of</td>
<td>Crafford and Hassan (2014) - the relationship between</td>
</tr>
<tr>
<td></td>
<td>indigenous fynbos.</td>
<td>elements of estuarine biodiversity and the recreational</td>
</tr>
<tr>
<td></td>
<td></td>
<td>fishery economy.</td>
</tr>
</tbody>
</table>

Source: Drawn from Barbier et al. (1997); King and Mazzotta (2000); Turner et al. (2008).

The production function method relies on knowledge of the relationship between the quality or quantity of the ecosystem service and the market output, or as described by Barbier (2007), the ecological-economic linkage underlying the service. Such relationships may not always be easy to
comprehend or measure (King and Mazzotta, 2000; Tuner et al., 2008). As noted by Barbier (2007),
the application of the production function approach is limited by an incomplete understanding of the
ecological functions and process underlying many ecosystem services and benefits.

3.4.2 Non-market valuation

Many ecosystem services and related benefits are not captured or valued within the current market
system, as such there is a gap between market valuation and the economic value of many ecosystem
services (Turner et al., 2008). To address this gap, several non-market valuation methods have been
developed and employed to value ecosystem benefits that are not reflected in the current market
system. Non-market valuation methods are used to estimate economic values for ecosystem services
and benefits where markets are non-existent or distorted. Non-market valuation methods include
revealed preference and stated preference methods. Revealed preference methods are based on
observed behaviour from which individual preferences are inferred (Heal et al., 2005). Stated
preference methods are based on responses to survey questions designed to elicit individual
preferences (Heal et al., 2005).

Revealed preference methods are used to value those ecosystem benefits that are accessed through
the consumption of market-priced private goods (Bateman et al., 2011), for example the travel and
time costs incurred in visiting a recreation site (King and Mazzotta, 2000). The demand for the
ecosystem service is derived from the demand for the market good directly linked to the ecosystem
service (Lange and Hassan, 2006); a substitutional or complementary relationship between the
ecosystem service and the marketed good (or goods) is assumed (Heal et al., 2005).

There are two common variants of the revealed preference approach and their application is limited to
a narrow range of ecosystem services (Heal et al., 2005). The travel cost method is mostly applied to
value ecosystem services related to recreation and tourism and hedonic pricing methods are
generally used to examine the effect of ecosystem attributes on property values (Barbier, 2007). The
less common averting behaviour method has been used to assess the willingness of individuals to
pay for health benefits (or avoid negative health effects) (Barbier, 2007) and is based on the premise
that individuals will invest money and change their behaviour to avoid undesirable health outcomes
(Heal et al., 2005). Heal et al. (2005) provide a discussion of revealed-preference methods and their
application in the context of aquatic ecosystem services. An overview of the more commonly applied
travel cost and hedonic pricing methods is given in Table 7.

| Table 7: An overview of revealed preference valuation methods |
|---------------|------------------|-------------------|
| Attribute | Travel cost | Hedonic pricing |
| Overview | Takes the travel and time costs incurred in visiting a recreation site as a proxy for recreational value; assumes that the value of the site or its recreational services is reflected in how much people are willing to pay to get there. | The demand for an ecosystem service is derived from the demand for a market good related to the ecosystem service. The basic premise being that the price of a marketed good is related to its characteristics, individual characteristics can be valued by looking at how the price people are willing to pay for the good change when the individual characteristic changes. |
| Suitable for | Ecosystem services related to recreation and tourism. | Estimating the economic benefits or costs associated with environmental quality (air, water or noise pollution) or environmental amenities (aesthetic views, proximity to recreational sites). |
| Value measure | Average value—measures total economic value, from which the average value for a day’s visit is often estimated. | Marginal value. |
| Value category | Direct and indirect use value. | |
### Data requirements

<table>
<thead>
<tr>
<th>Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trip and visitor information collected through a survey of visitors; data on explanatory variables that are likely to influence visit rates (income, preference and availability of alternative sites).</td>
<td>Large amounts of data on the market price (e.g., property prices) and characteristics of the good. Data may be expert opinion of property values, self-reporting or related to actual sales (most accurate).</td>
</tr>
</tbody>
</table>

### Strengths

<table>
<thead>
<tr>
<th>Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimated values are revealed from actual behaviour of individuals.</td>
<td>Relatively inexpensive to apply if data are readily available.</td>
</tr>
</tbody>
</table>

### Challenges

<table>
<thead>
<tr>
<th>Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Relies on variation within travel costs or variations in ecosystem quality within a single (property) market.</td>
<td>Difficulties arise when trips are to more than one destination or for more than one purpose; limited to recreational benefits; remains controversy over whether to include visitor’s travel time and how to value it; surveys can introduce sampling biases; relating recreational quality to environmental quality can be difficult; provides information about current conditions, but not about gains or losses from anticipated changes in resource conditions.</td>
</tr>
</tbody>
</table>

### Example

<table>
<thead>
<tr>
<th>Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Used to estimate the in situ value of visiting a recreational site; used to value recreational fishing and beach use.</td>
<td>Most commonly applied to variations in housing prices that reflect the value of local environmental attributes; used in valuing air quality, flooding, some use in valuing water quality.</td>
</tr>
</tbody>
</table>

### Application in SA

<table>
<thead>
<tr>
<th>Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Study of the value of recreational benefits associated with averting river water inflow reductions to the uMgeni Estuary, KZN (Chege, 2009; Hosking, 2011). Direct use value of an estuary was estimated by examining actual expenditure by holiday makers (Turpie et al., 2009).</td>
<td>Used to determine the effects of an estuary on property values – to measure direct use value of the estuary (Turpie et al., 2009).</td>
</tr>
</tbody>
</table>

Source: Drawn from King and Mazzotta (2000); Pagiola et al. (2004); Heal et al. (2005); Lange and Hassan (2006); Turner et al. (2008).

Stated preference methods - contingent valuation and choice experiment techniques – involve surveying individuals regarding the benefits of an ecosystem, ecosystem service or range of services to elicit their willingness to pay for the attribute under consideration (Barbier, 2007). Individuals are presented with a hypothetical scenario(s) involving an ecological attribute and asked directly about their WTP (contingent valuation) or to make choices between various options (choice modelling) (King and Mazzotta, 2000). The techniques differ in how the ‘change’ to be valued is described (effect on ecosystem service or effect on attributes to be valued) and the format of the survey response (direct statement of WTP or choice between options) (Heal et al., 2005). Whereas many of the other valuation methods are more specific to a particular type of ecosystem service or value (i.e. use or non-use value, see Section 3.2.3), stated preference methods have the potential to be used more widely across services and value categories (Heal et al., 2005). An overview of stated preference valuation methods is given in Table 8.

Heal et al. (2005) note that in order to apply a stated preference method the change in the ecosystem under consideration must be described in the survey in terms of attributes that individuals value and in such a way that individuals can understand the valuation scenario. King and Mazzotta (2000) emphasise that the survey should focus on specific services or benefits (rather than a broad change ecosystem condition) and that the context must be clearly defined. As such, careful attention must be given to the survey design (including pre-testing) which may be a lengthy and expensive exercise (King and Mazzotta, 2000). Barbier (2007) indicate the potential difficulty in describing how changes

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14 Choice experiment methods include a variety of techniques such as choice experiments, contingent ranking, contingent rating and paired comparisons (Turner et al., 2008) and originate in conjoint analysis (Boxall et al., 1996)
in ecosystem processes and components affect ecosystem services and, in turn, how these changes affect the benefits or attributes valued by individuals.

**Table 8: An overview of stated preference valuation methods**

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Contingent valuation</th>
<th>Choice modelling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overview</td>
<td>Individuals are surveyed to elicit their preferences, in the form of statements, ratings, rankings or choices, for predefined (hypothetical) alternatives regarding changes in ecosystem services/benefits.</td>
<td>Respondents are asked to state, or are asked a question that will reveal, their willingness to pay (WTP) to gain an improvement, or avoid a detrimental change, in the provision of a good or service. Alternatively (or additionally) they may be asked what they are willing to accept (WTA) to forgo an improvement or tolerate a detrimental change.</td>
</tr>
</tbody>
</table>

**Suitable for** | A broad range of ecosystem services. |
**Value measure** | Marginal, average and total value can be estimated depending on purpose of study. |
**Value category** | Use and non-use values, only method for eliciting non-use values. |
**Data requirements** | Design survey instrument to capture all data required for estimating values, survey sample must be representative of the population of interest, econometric expertise and software to analyse responses and estimate WTP/WTA functions, as well as validity and reliability testing. |

**Strengths**
- Can be used to value the outcomes of an action as a whole, as well as various attributes of the action; allows respondents to think in terms of trade-offs, which may be easier than directly expressing monetary values.
- Can be used to value the outcomes of an action as a whole, as well as various attributes of the action; allows respondents to think in terms of trade-offs, which may be easier than directly expressing monetary values.
- Assumptions individuals understand the service/benefit in question and will reveal their preferences in the hypothetical market just as they would in a real market; vulnerable to respondent bias; can be expensive and time-consuming, because of the survey design and extensive pre-testing; scepticism (across decision-makers, individuals, society) of the results; based on asking people questions, as opposed to observing their actual behaviour (which is a source of much controversy).

**Challenges**
- Not possible to analyse the attributes of the change in question without designing different valuation scenarios for each level of the attribute.
- As the number of attributes (or choices) increases, inconsistency in responses likely to increase as respondents lose interest or become frustrated; translating responses into money values may increase uncertainty.
- Measure the use and non-use values that citizens have for parks/conserved land; estimate how much visitors to a waterfall site would be willing to pay for increased overflows; consider the value of dam recreation compared to the market value of hydropower supply.
- Investigate the social and economic trade-offs and values associated with the location of a landfill – survey asked residents to choose between pairs of hypothetical sites and locations for a new landfill; estimate the relative preferences of residents for conserving/restoring an environmental resource.

**Example**

| Application in SA | Hosking (2010, 2011) to investigate the value of recreational benefits associated with averting river water inflow reductions to estuaries. | Turpie and Joubert (2001) to examine the response in tourism value to changes in river quality. |

Source: Drawn from Boxall et al. (1996); King and Mazzotta (2000); Pagiola et al. (2004); Turner et al. (2008).

While being widely applied (Heal et al., 2005), much debate remains around the application of stated-preference methods and the credibility of the estimated values (Heal et al., 2005; Turner et al., 2008). Bateman et al. (2011:192) suggest that stated preference methods are appropriate when individuals are familiar with ecosystem service/benefit and have “clear prior preferences for the goods in question or can discover economically consistent preferences within the course of the survey exercise”. Turpie and Kleynhans (2010) draw attention to the additional challenge of applying stated preference
methods in a rural, developing country context and note the importance of the survey design and implementation.

3.4.3 Cost-based methods

Cost-based methods, sometimes included under market-based methods (Turner et al., 2008; Pascual et al., 2010) or revealed preference methods (Pagiola et al., 2004) are considered here as a separate category (following Grossman, 2012). Cost-based methods use various costs as a proxy for benefits and include methods based on the costs of avoiding damages due to lost services and the costs of replacing services either through restoring the ecosystem providing the services or through providing substitute services (King and Mazzotta, 2000). An overview of methods is provided in Table 9. A widely applied approach is to use the cost of treating water to represent the benefit of water filtration services provided by wetlands (Pagiola et al., 2004). Cost-based methods are also used in the field of health economics, where a ‘cost of illness’ approach (all the costs of treating a patient with a particular illness) is used as indication of the benefit to the patient of avoiding the illness (Barbier, 2007).

Strictly, replacement costs are not a measure of economic value as the method is not based on individual preferences and does not measure an individual’s willingness to pay for a service (King and Mazzotta, 2000; Heal et al., 2005). The key assumption of cost-based approaches is that “if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them” (King and Mazzotta, 2000: section 5). Pagiola et al. (2004) (citing Shabman and Batie, 1978) note three conditions that must hold for the approach to be valid: (1) the replacement service must be equivalent in quality and magnitude to the ecosystem service (perfect substitute); (2) the replacement must be the least cost option of replacing the service; and (3) people would actually be willing to pay the replacement cost to obtain the service (there would be a demand). Where the conditions do not hold, cost-based approaches are unlikely to accurately reflect the benefits of ecosystem services (Pagiola et al., 2004; Heal et al., 2005; Turner et al., 2008).

Cost-based methods can most appropriately be applied where damage avoidance or replacement expenditures have actually occurred (King and Mazzotta, 2000) in which case the cost-based estimate would reflect a lower-bound of the benefit of the value (Brouwer et al., 2013). The use of replacement cost methods for valuing ecosystem services and benefits is often a response to the frequent lack of information on the links between ecological function, processes and components and the ecosystem services that benefit humans (Barbier, 2007), and how changes in the underlying condition of the ecosystem influence the delivery and use of ecosystem services. Without such information, it is difficult to construct reliable scenarios through which to elicit individual preferences regarding changes in ecosystems and ecosystem services (Barbier, 2007).

A well-known example of the application of the replacement cost approach was the decision to restore the Catskills catchment for the specific purpose of providing clean drinking water to New York City based on a comparison of the costs of protecting and restoring the catchment to replacing the water purification services of the catchment with a new drinking water treatment facility (Heal et al., 2005). Cost estimates indicated that building and operating the water treatment system would be significantly greater than protecting and restoring the catchment and New York City chose to protect the Catskills. Barbier (2007) notes that it was sufficient for the policy decision simply to demonstrate the cost-effectiveness of the restoration and protection option compared to the alternative.
### Table 9: An overview of cost-based valuation methods

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Damage costs avoided</th>
<th>Replacement, restoration and substitute costs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overview</td>
<td>Costs are used as a proxy for the value of benefits derived from ecosystems.</td>
<td>Costs of replacing the benefits of the ecosystem are used to value the benefit.</td>
</tr>
<tr>
<td>Suitable for</td>
<td>Cases where damage avoidance or replacement expenditures have actually occurred, specific conditions must hold. Where only a single service is being considered.</td>
<td></td>
</tr>
<tr>
<td>Value measure</td>
<td>Lower bound estimate of benefit; proxy of marginal or average value depending on the study.</td>
<td>Net average value based on market price of replacement; can be used as proxy for marginal value.</td>
</tr>
<tr>
<td>Value estimated</td>
<td>Costs of alternative approaches; comparisons can be compared to determine cost-effectiveness.</td>
<td></td>
</tr>
<tr>
<td>Data requirements</td>
<td>Data on the change in an ecosystem (and service) of interest and associated substitution effects. Relate changes in the asset (ecosystem) to the probability of the damage event occurring.</td>
<td>Ascertained service loss, estimate cost of restoring ecosystem to provide the original level of service or cost of replacing the service or of substitute services. Information on replacement costs can be obtained from direct observation of actual spending on substitutes or from engineering estimates of restoration costs.</td>
</tr>
<tr>
<td>Strengths</td>
<td>Less data and resource intensive – it is often easier to measure the costs of producing benefits rather than the benefits themselves; data or resource limitations may rule out valuation methods that estimate willingness to pay; provide a rough indication of economic value, subject to the degree of substitutability between related services; improved validity when based on actual expenditure.</td>
<td></td>
</tr>
<tr>
<td>Challenges</td>
<td>If conditions (substitutability, demand for substitute, lowest cost alternative) do not hold, method likely to over or under-estimate actual value. Few environmental resources have such direct or indirect substitutes. Substitute goods are unlikely to provide the same types of benefits as the natural resource. Should be used with caution. Not a measure of economic value as the method is not based on individual preferences and does not consider individuals’ behaviour in the absence of services.</td>
<td>Assumes the costs of replacement equal the original benefits, original benefits may exceed replacement costs and method will underestimate value of benefits. Alternatively, additional benefits of the substitute may exceed the original benefit, and overestimate the value of the ‘lost’ benefit. Assumes complete replacement/ restoration is feasible.</td>
</tr>
<tr>
<td>Example</td>
<td>If a wetland protects adjacent property from flooding, the flood protection benefits may be estimated by the damages avoided if the flooding does not occur or by the expenditures property owners make to protect their property from flooding.</td>
<td>Valuing improved water quality by measuring the cost of controlling effluent emissions; valuing erosion protection services of a forest or wetland by measuring the cost of removing eroded sediment from downstream areas; valuing the water purification services of a wetland by measuring the cost of filtering and chemically treating water; valuing fish habitat and nursery services by measuring the cost of fish breeding and stocking programs.</td>
</tr>
<tr>
<td>Application in SA</td>
<td></td>
<td>de Lange et al. (2012) - cost of illness approach to estimate the impacts of microbial pollution.</td>
</tr>
</tbody>
</table>

Source: Drawn from Barbier et al. (1997); King and Mazzotta (2000); Pagiola (2004); Heal et al. (2005); Barbier (2007); Turner et al. (2008).
Heal et al. (2005) suggest that the Catskills example fulfils the conditions set out by Shabman and Batie. Barbier (2007) further suggest that, while a stated preference method may have elicited an estimate of the total willingness-to-pay by New York City residents for the quantity of freshwater provided, eliciting an estimate of the willingness-to-pay to avoid a decline in the water treatment service as a result of land-use changes in the catchment that affect the free provision of the service would have been a significantly greater challenge.

3.4.4 Benefits transfer

The benefits transfer method involves the ‘transfer’ of primary valuation estimates in one context to estimate values in a different context (Pagiola et al., 2004). For example, the value of wildlife viewing estimated for a specific reserve may be used to estimate the value of wildlife viewing at a different reserve (Pagiola et al., 2004). Benefit estimates may be transferred (1) directly as value estimates; (2) as value estimates adjusted for the new site/context; or (3) as the functional form (equation) estimated in the primary valuation (Heal et al., 2005; Turner et al., 2008).

Given the context specific nature of ecosystem service valuation, the transfer of value estimates from one context to another may be characterised by high uncertainty and it is preferable to conduct primary assessments if resources allow (Brouwer et al., 2013). The benefit transfer method is generally applied when time and resources are not available for primary site specific valuation (King and Mazzotta, 2000). Heal et al. (2005) note the use of this method in policy analyses. The accuracy of the benefit transfer method depends on the availability and reliability of primary valuation results (King and Mazzotta, 2000; Brouwer et al., 2013) and may be limited by the availability of suitable primary studies on which to base the benefit transfer (EPA, 2014). With an increase in the number of reliable primary valuations, the scope for the application of the benefit transfer method will also increase (Brouwer et al., 2013).

While much debate remains on the use of the benefit transfer method, Pagiola et al. (2004) note that there is some consensus that the approach can provide meaningful and valid results under certain conditions. The reliability of the method is improved when the characteristics, such as ecosystem quality, location, and affected population, of the original site and the study site are similar; the ecological change (and its implications) are similar for the two sites; and when the primary valuation study was rigorous and used appropriate valuation techniques (King and Mazzotta, 2000).

Pearce and Seccombe-Hett (2000) use a diagram to summarize the various economic valuation methods and link them to the broad value category to which they are suitable, Figure 4. Under this categorisation, the production function approach is not considered as a specific valuation method itself, but rather as a part of several other methods; and cost-based methods are not included (Pearce and Seccombe-Hett, 2000).
3.4.5 Supporting tools and approaches

There is still much debate about how to derive ecosystem values, both within and outside the economics discipline. In recent years, the valuation of ecosystems has increased and new tools and approaches have emerged as analysts try and overcome some of the limitations and assumptions of strict neoclassical economics (Gómez-Baggethun and Muradian, 2015). In addition, there are many non-economic approaches\(^{15}\) to expressing the value of ecosystems and, increasingly, valuation methods stemming from different disciplines are used together or in support of one another. For example, Bunse et al. (2015) and Lienhoop et al. (2015) investigate the combination of deliberative processes and economic valuation. Additional methods and tools of ecosystem valuation include:

- Discourse, participatory and deliberative approaches - focus groups, citizens’ jury (Wilson and Howarth, 2002; Christie et al., 2012);
- Bayesian networks (McVittie et al., 2015);
- Deliberative monetary valuation – valuation workshops and market stalls (Lienhoop et al., 2015);
- Spatial representation of ecosystem services (Burkhard et al., 2012), landscape-scale spatial analysis (Castro et al., 2014);
- Evidence-based narratives (Davis et al., 2015);
- Specifically developed valuation models such as the computer-based Integrated Valuation of Ecosystem Services and Tradeoffs (INVEST) tool (Tallis and Polasky, 2009); and
- Assessment guidelines such as the Toolkit for Ecosystem Service Site-based Assessment (TESSA) (Peh et al., 2013);

\[^{15}\text{For introductions to non-economic methods of valuation see eftec (2006); EPA (2009) and TEEB (2010).}\]
3.5 Use of ecosystem valuation information

Many intended uses for the information generated through ecosystem valuations have been suggested. Through a review of the literature, Laurens et al. (2013) identified three broad categories of potential use:

- Informative - to raise awareness or advocate a particular course of action;
- Decisive - to inform a specific decision, such as prioritizing ecosystem intervention efforts; and
- Technical - to inform the design of an economic instrument, for example setting prices for a payment for ecosystem services (PES) scheme, damage compensation.

The sourcing and securing of funding for ecosystem interventions is an additional use of valuation information (van Zyl et al., 2004; van Beukering et al., 2007; Verdone, 2015).

Pearce and Seccombe-Hett (2000) describe several specific uses, the relevant category proposed by Laurens et al. (2013) is given in brackets:

- Cost-Benefit Analysis (CBA) of Projects (Decisive);
- CBA of Policies (Decisive);
- Pricing Policy (Technical);
- Design of Environmental Taxes (Technical);
- National Accounting (Informative) – incorporating environmental assets and ecosystem services into a country’s national accounts
- As a Management Tool (Decisive) – Pearce and Seccombe-Hett (2000) suggest that economic valuation can play a role in environmental management by providing an indication of the willingness to pay (WTP) for different attributes of a given ecological asset. The asset or ecological infrastructure can then be managed to secure and improve those attributes that attract the greatest WTP. This type of use is less well explored relative to more conventional costs-benefit analysis;
- As a Participatory Exercise (Decisive) – certain valuation techniques involve direct interaction with people regarding their preferences for or against changes in ecosystems and ecosystem services. Through discussion and debate, this process can lead to decisions (final changes) that are acceptable to affected parties (Pearce and Seccombe-Hett, 2000).

Through a review of coastal and marine valuation studies, Waite et al. (2014) provide a useful summary of cases where valuation information has been used in making a specific decision, a portion of these studies are reported in Appendix A and related to the ‘use categories’ proposed by Laurans et al. (2013) as illustrative examples.

Through their review of the use of valuation information, Laurans et al. (2013:214) found a “paucity of papers that describe, through a case study, how a specific ESV [ecosystem service valuation] has played a role in a decision”. The authors attribute the finding to several possible factors including challenges of the review itself, but suggest that elements of uncertainty regarding the accuracy of value estimates, insufficient capacity of decision-makers to make use of value estimates, and the restrictive cost of valuation assessments may limit the use of valuation information in decision-making. Bockstael et al. (2000) and Pagiola (2004) contend that the usefulness of economic valuation information has been limited by poorly defined ‘value’ questions and a failure to frame the valuation analysis to answer a specific question. In the context of water quality valuation, Keeler et al. (2012) note that valuation assessments seldom satisfy the expectations of decision makers. Laurens et al. (2013) highlight that there is still much debate on the role of ecosystem valuation in promoting ecosystem conservation and management, with several arguments suggesting that ecosystem valuation is “neither necessary nor sufficient for coherent and consistent choices about the environment” (Laurans et al., 2013 citing Vatn and Bromley, 1994). Primmer et al. (2015) argue that comparatively little attention has been given to how ecosystem management decisions are made and
to which arguments are effective in promoting ecosystem conservation. The authors warn against the assumption that decisions will change as new information becomes available, suggesting that this assumption disregards the role governance systems play in influencing decisions and policy implementation. Primmer et al. (2015) advocate for research on the governance of ecosystems, how ecosystem management decisions are made and implemented and which arguments influence these processes.

4 Principles and Approach to Ecosystem Valuation

In the international literature, various discussions, reviews and guides to valuing the benefits of ecosystems are available (introductions are provided in Barbier et al., 1997; Pagiola et al., 2004; Heal et al., 2005; etf, 2006; Barbier, 2007; Turner et al., 2008; EPA, 2009; TEEB, 2010; DEFRA, 2011; for restoration specific examples see Rutherford et al., 2000; Lewis III, 2001; Pendleton and Balder, 2010; Robbins and Daniels, 2012). Empirical studies, particularly of the value of a change in the system as a result of an action, are less common (Rey-Benayas et al., 2009; Wortley et al., 2013). The estimates that emerge from an economic valuation assessment will be specific to the context of the assessment and will be influenced by the human preferences, institutional arrangements and the cultural setting in which the valuation takes place (Barbier et al., 2009:249). Furthermore, the valuation approach and methods applied will influence which values are emphasised and how values are expressed (Vatn, 2005).

The economic valuation of changes in the condition of ecological infrastructure is concerned with well-defined changes in the ecosystem(s) and the changes in human well-being that result (Bockstael et al. 2000; Pagiola et al. 2004; Pendleton and Balder, 2010). Valuing the benefits of investments in ecological infrastructure depends on identifying and quantifying a change in ecosystem service and benefit as a result of the investment (Pendleton and Balder, 2010). DEFRA (2011) capture this relationship in an impact pathway of policy change, Figure 5.

![Figure 5: Impact pathway of policy change](image)


Keeler et al. (2012) apply a similar approach to the specific context of water quality, Figure 6. Such frameworks are useful is conceptualizing the ‘chains of causality’ between an action or investment affecting ecosystem condition and the creation of value (not necessarily limited to economic value). These frameworks emphasise that the value of the action or investment stems from the resulting changes in the ecosystem and the services and benefits delivered. In an economic value sense, for these changes to have value, they must be ‘preferred’ by at least one individual.

![Figure 6: Framework linking actions to values for water quality related ecosystem services](image)

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16 A term borrowed from Ginsburg et al. (2010).
From a review of the literature and particularly an analysis of 18 valuation frameworks or guidelines (see Appendix B for a table of references) several elements of the valuation process common across the literature were identified: (1) specification of the study question; (2) biophysical assessment of the ecosystem and ecosystem services (prediction/quantification of service provision and change); and (3) valuation of the changes in services and benefits. Four stages of the valuation process and aspects for consideration at each stage are discussed in the following sections.

**Specification of the study boundaries and context**

The goal of the economic assessment should be to generate information relevant to the decision-maker, stakeholder or affected party. For economic valuation to provide meaningful, relevant information, the assessment must be a response to a specific question or information need and have an intended purpose as ascertained through engaging with the relevant decision-maker, stakeholder or community (Bockstael et al., 2000; Pagiola et al., 2004; Ervin et al. 2014; Waite et al., 2014). The valuation must be clearly linked to a specific action (e.g. land-use change, ecosystem restoration) that results in a change in the ecosystem and delivery of ecosystem services (Keeler et al., 2012). Rather than estimating the total value of an ecosystem, the value of a change in the ecosystem or ecosystem service is of interest (Bockstael et al. 2000; Turner et al., 2003; Pagiola et al. 2004; Heal et al., 2005; Fisher et al., 2008; Pendleton and Baldera, 2010; Bateman et al., 2011; DEFRA, 2011; Vieira et al., 2014; Verdone et al., 2015); in economics this is referred to as the marginal value of the ecosystem service (see Section 3.2.4 for a discussion of marginal value). The baseline or reference point from which the change is evaluated is of importance (Heal et al., 2005).

The extent to which ecosystem services and benefits are considered ‘valuable’ will be influenced by the physical, economic, cultural and institutional setting in which the valuation takes place (Turner et al., 2008; Barbier et al., 2009; Keeler et al., 2012). Ecosystem benefit values are context specific. Gómez-Baggethun et al. (2014:8) argue that the “context dependency of values...has been under-communicated in the discourse on valuing ecosystem services”. Considering the socio-ecological context of the study area is an important component of the valuation process and will assist in identifying key ecosystem services, dependencies and beneficiaries and provide insight into the perspectives of the users of the ecosystem (Haines-Young and Potschin, 2009). In placing the valuation assessment in context, drivers of change such as physical drivers (climate change), socioeconomic drivers (affecting demographic structure and market forces) and policy changes should be considered (Mace et al., 2011). The boundaries of the study, in terms of the intended intervention options, geographical and temporal scale and the available resources (data, time, funds) must be identified and clearly defined (Waite et al., 2014). The information gathered should be used to interrogate whether a valuation study is feasible, ethically appropriate, and relevant given the objectives, study context and available resources.

The valuation study should be designed or framed in such a way as to provide the desired information. In summary, Heal et al. (2005) suggest four key elements of ‘framing the valuation’ question:

- The likely changes in ecosystem services brought about by the decision alternatives;
- The intended scope of the analysis (ecosystem functions or services and types of value);
- The spatial scale (geographic extent and definition of the relevant population);
- The temporal scale.

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17 A useful discussion of drivers of ecosystem change is provided in chapter 3 of the UK National Ecosystem Assessment (Winn et al., 2011).
System conceptualization

The ecological response to the action needs to be translated into or related to attributes that society values. As noted by Keeler et al. (2012), the outputs of biophysical models (such as nitrate concentrations in water) are seldom the attributes directly valued by people (such as safe water for drinking or recreation). The ecological responses must be clearly linked to a resulting impact on human well-being (which can then be valued) (Jenkins et al., 2010; DEFRA, 2011; Vieira et al., 2014).

A conceptual model of how the system works and what is expected to change with the action is useful for identifying and translating the ecological response into human well-being impacts (EPA, 2009). It can assist in identifying the services/benefits and affected parties most relevant to the value question and context; inform the selection of appropriate valuation methods; and identify modelling/monitoring requirements to provide reliable data on which to build the assessment. Ginsburg et al. (2010) describe this process as formulating the causal chains linking an action or management scenario to the resulting change in ecosystem assets and the effect on ecosystem services. Ginsburg et al. (2010) suggest a ‘Comparative Risk Assessment’ as a useful mechanism for developing ‘chains of causality’. Similarly, Hackett (2006) suggest that risk assessments could be used as a starting point for ecosystem valuation. The key principle is that ecosystem valuation is an interdisciplinary exercise (Haines-Young and Potschin, 2009; Ginsburg et al., 2010; Brouwer and Georgiou, 2012) requiring inputs from the social, economic and biophysical disciplines. As explained by Haines-Young and Potschin (2009:55), “Biophysical assessments are needed to provide an understanding of how services are generated; socially grounded economic analysis is required to estimate the relative worth of services”.

The study contextualization and system analysis processes should be used to identify sources of value derived from the ecosystem and establish which ecosystem services and benefits are most likely to change with the action. The groups of people who are likely to be affected by the outcomes of the action can then be identified. Changes in the delivery of ecosystem services are likely to have different consequences across various groups of society, changes may be beneficial to some, while negatively affecting others.

Ecological assessment

The change in the delivery of ecosystem services must be quantified (either measured, modelled, or predicted through expert judgement). The ecological response to an action or investment may increase the supply of some services while decreasing others, or may prevent a future loss in service supply rather than increasing supply (i.e. by halting further degradation wetland rehabilitation may maintain the current supply of services) (Pendleton and Baldera, 2010). A comparison of the ‘before and after’ situation may not reveal the value of the action; a ‘with or without’ comparison is necessary.

To be able to demonstrate that it is the ecological outcomes of the action that have created value, the effects of the action must be isolated from other factors driving the supply and/or use of ecosystem services and benefits (Pendleton and Baldera, 2010). This is an important principle when designing the data collection process. Considering the context of the study and developing a rigorous conceptual model of the system can assist in identifying potential external drivers of change (climate change, national and international policy, management practices) and factors that influence the demand for ecosystem services and benefits (such as population size, income level, availability of alternatives and complements, preferences).

Valuation

There are many methods and tools available for valuing ecosystem services and benefits; there is no single method with greater validity and no single method that captures all values (Heal et al., 2005). For each valuation, the combination of methods (including non-economic approaches) appropriate to
the context of the specific assessment must be chosen (eftec, 2006; DEFRA, 2011). The selection of methods will depend on several factors, including:

- The decision question and intended purpose and audience of the valuation information;
- The nature of the ecological change and expected effect on ecosystem services and benefits and the level of uncertainty regarding the expected change (some methods perform better under uncertainty than others);
- The scope and extent of the study (which services and beneficiaries will be considered; how values are to be expressed; the geographical and temporal scope);
- Availability of data and supporting information;
- The inherent characteristics of the valuation methods and the type of output they generate; and
- The resources (budget, time, human capacity) available (Heal et al., 2005; eftec, 2006; Hitzhusen, 2007; van Beukering et al., 2007; Turpie and Kleynhans, 2010).

Economic models should be used with biophysical models of ecosystem and service delivery changes (EPA, 2009; DEFRA, 2011). Two types of data are fundamental to the valuation process:

- Ecological (biophysical) data – in determining the change in supply of services and benefits (quantity and quality changes); and
- Economic outcome data – in determining the use and willingness to pay for the changes in services and benefits (Pendleton and Baldera 2010).

The valuation should consider the multiple benefits and/or costs of a change in the ecosystem, its attributes and the delivery of services (Keeler et al., 2012). For example, changes in water quality may influence drinking water treatment, recreational water use, irrigation water use and have consequences for human health. However, not all ecosystem services are compatible (i.e. both services cannot occur simultaneously) and the values of the benefits derived from these services cannot be added together (DEFRA, 2011).

The distribution (across demographic groups, space and time) of the costs and benefits associated with the action must be considered. The assessment should identify who gains and who loses across affected stakeholder groups. Changes in the ecosystem may be beneficial to certain groups, while being detrimental to others. Similarly, benefits gained in one location (geographically or temporally) may result in costs elsewhere. An un-even distribution of costs and benefits can have both practical and ethical consequences. Ethically, the analysis of the distribution of costs and benefits is important to ensure that interventions do not harm vulnerable people or lead to social exclusion (Pagiola et al. 2004). Practically, the costs and benefits received by local users can have a strong influence on how the ecosystem is used and managed and therefore the success and longevity of the action/investment (Pagiola et al. 2004).

Economic valuation should be based on the best scientific information available; sources of uncertainty should be identified and the limitations of the valuation must be acknowledged (EPA, 2009; Pascual et al., 2010; Brouwer & Georgiou, 2012; Ervin et al., 2014). Conservative assumptions, values and methodologies should be used when uncertainty is great (WBCSD; 2011). Potential biases should be identified and reduced wherever possible (WBCSD, 2011). Clear and sufficient information should be provided for reviewers to assess the credibility and reliability of the assessment (WBCSD, 2011) and to encourage professional and stakeholder scrutiny (Ginsburg et al., 2012).

The valuation must be transparent. All assumptions and expert judgements must be clear, including the basic assumptions of the economic theory and models applied. Each valuation assessment is context-specific; the theoretical rationale behind the chosen approach is important and should be
explained (La Notte et al., 2015). The assessment must identify and be explicit about which values are captured through the valuation and which are excluded (Heal et al., 2005).

The value information should be communicated to the intended audience and to interested and affected members of the public. Findings and methods should be shared with the broader research community to encourage improvement in valuation approaches and tools (Ginsburg et al., 2012). Attention should be given to communicating the outcomes in such a way that the valuation information is accessible to the intended audience and misinterpretation is reduced (EPA, 2009; Ervin et al., 2014).

4.1 Challenges and Opportunities

Ecosystem valuation is an evolving field requiring an interdisciplinary effort and drawing from various knowledge systems (Gómez-Baggethun et al., 2014). As such there are numerous debates concerning both theoretical and operational aspects of ecosystem valuation. Liu et al. (2010) introduce various debates on the use of ecosystem valuation information, covering ethical and philosophical, political and methodological debates. Similarly, Farley (2012) provide an overview of the ‘economics debate’ covering issues around sustainability, justice and the goal of efficiency. Turner et al. (2003), Pagiola et al. (2004) and DEFRA (2011) highlight challenges related to the application of ecosystem service valuation. Chapter 6 of The Economics of Ecosystems and Biodiversity (2010) is dedicated to issues of discounting (Gowdy et al., 2010); in the South African context the selection of a discount rate is discussed in Hosking et al. (2002), van Zyl & Leiman (2002), van Zyl et al. (2004), Blignaut et al. (2013b) and Mullins et al. (2014). Ethical debates are discussed in Luck et al. (2012) and Jax et al. (2013). Christie et al. (2012) draw attention to the specific challenges of valuing biodiversity (through monetary and non-monetary approaches) in less developed countries and suggest solutions where possible. Pendleton and Baldera (2010) discuss issues of monitoring and data collection with regards to the economic value of habitat restoration. In the following section, the two related challenges of addressing threshold effects and valuation under uncertainty are discussed in greater detail.

4.1.1 Threshold effects

Economic valuation captures the value of a relatively small change in ecosystem service and benefit supply as a result of a change in the ecosystem over some non-critical range (Turner et al., 2003; Farley, 2012). Beyond a non-critical range or at an ecological threshold18, any (even small) change in ecosystem condition can lead to an abrupt and substantial change in the state of the system (Farley, 2012) resulting in a significant disruption in the provision of ecosystem services (Heal et al., 2005; Crépin et al., 2012). Under such large-scale changes economic valuation methods are likely to be less robust (Pagiola et al., 2004; DEFRA, 2011) and marginal concepts of ecosystem valuation are unlikely to capture the effect of threshold level changes in ecosystem service provision (Heal et al., 2005).

Whether a change in ecosystem service and benefit provision is gradual (marginal) or drastic is an important consideration in the valuation process. Application of the marginal concept of value under conditions of large-scale ecosystem service change may not be appropriate or may need to be addressed in a particular way (Barbier, 2007; DEFRA, 2011). In some cases, it may be possible to identify threshold points and predict when they may be reached, for example in the transition from oligotrophic to eutrophic states of water bodies (Heal et al., 2005), in others the difficulty of predicting

18 Anderson et al. (2009) define an ecological threshold: as “the critical value of an environmental driver for which small changes can produce an ecological regime shift” and an ecological regime shift as “a sudden shift in ecosystem status caused by passing a threshold where core ecosystem functions, structures and processes are fundamentally changed”. However, there are multiple definitions and terminologies (Jax, 2014).
or even identifying ecological thresholds is much greater (Barbier, 2007, Jax, 2014). Crépin et al. (2012) provide a discussion on the theory of ecological thresholds and regime shifts. Fisher et al. (2008) elaborate on marginal analysis and the need to identify a ‘safe minimum standard’ zone. Anderson et al. (2009) review statistical techniques with the potential to support the identification of ecological thresholds and regime shifts. Farley (2012) suggest that in certain cases it may be impossible to predict where an ecological threshold lies, or when it may be reached.

Farber et al. (2002:386) make the distinction between ecological and economic thresholds, suggesting that “critical thresholds in ecosystem structure or function do not necessarily imply economic thresholds for values… the opposite may be true also—i.e. gradual changes in natural conditions may lead to non-linear changes in economic conditions”; the authors provide the example of a lake ‘suddenly’ being closed to swimmers due to a gradual decline in water quality (rather than an abrupt, drastic decline in water quality). Such distinctions relate to how threshold and reference levels are determined; whether they are defined in terms of their potential to provide services to humans or not (Jax, 2014). Where ecological conditions and threshold levels are uncertain, ecosystem valuation is undertaken under uncertainty. Dealing with non-linear relationships, such as the specific case of ecological thresholds, presents both a challenge and opportunity for ecosystem valuation research.

4.1.2 Valuation under uncertainty

There is considerable uncertainty associated with both the functioning of ecosystems and the valuation of their attributes, as such dealing with uncertainty is an important consideration in ecosystem valuation. Uncertainty may arise at any stage of the valuation process and be related to incomplete knowledge of ecosystem functioning and the relationship between ecosystems and human well-being (i.e. the supply of ecosystem services) (Turner et al., 2008); predictions regarding the ecological and economic conditions of the future (Turner et al., 2008); assumptions about the way individual preferences regarding ecosystem services are formed (preference uncertainty) (Pascual et al., 2010); and the application of valuation methods (technical uncertainty) (Heal et al., 2005; Pascual et al., 2010).

Two variants of uncertainty or imperfect knowledge can be distinguished, risk and uncertainty. Risk encompasses those cases where “the possible consequences of a decision can be completely enumerated, and probabilities assigned to each possibility” (Perman et al., 2003:445). Uncertainty refers to the case where it is not possible to assign probabilities to all possibilities (also referred to as ‘states’ or ‘outcomes’) (Perman et al., 2003). Under conditions of risk, meaningful probabilities can be assigned to likely outcomes; under conditions of uncertainty, probabilities are unknown (Turner et al., 2008). A further distinction can be made between ‘uncertainty’ and ‘radical uncertainty’, the former referring to the situation where all possibilities (states) can be specified, but probabilities cannot be assigned; and the latter where all possibilities cannot be specified (Perman et al. 2003). Detailed introductions to the various sources of uncertainty in ecosystem valuation and options for addressing risk and uncertainty are provided by EPA (2009) and Pascual et al., (2010).

In the case of ecosystem valuation under risk, risk can be expressed in terms of the likelihood of a particular outcome. Each potential outcome is weighted by the probability (likelihood) of its occurrence (Pascual et al., 2010). Probability distributions require “statistical data that are sufficiently extensive to allow some confidence in their predictions” (Heal et al., 2005:216). A relatively simple approach to probabilistic uncertainty analysis is sensitivity analysis, where each parameter or assumption is varied at a time to produce a range of estimates of the value under consideration (EPA, 2009). The likelihood of occurrence (or probabilities) are not calculated for the estimated values; as such each value in the range could be interpreted as being equally likely to be the outcome. Due to this simplification and the complexity of ecosystems, simple sensitivity analysis may be potentially misleading (EPA, 2009). An alternative to sensitivity analysis is Monte Carlo analysis which allows
multiple sources of uncertainty to be considered simultaneously and generates probabilities for each value within the estimated range (EPA, 2009). A South African example of this approach is the study by Crookes et al. (2013) who employed a risk analysis framework based on Monte Carlo simulation to assign a probability distribution to the outcomes of several ecological restoration projects undertaken in South Africa.

Heal et al. (2005) note, that often in the context of ecosystem valuation, data for probabilistic uncertainty analysis is not always available. Assigning ‘subjective’ probabilities based on expert knowledge is one way to address risk where it is not possible to assign objective probabilities through data analysis (Heal et al., 2005). In assigning subjective probabilities, experts provide judgments about the likelihood of different outcomes. A variety of expert elicitation methods are available and may involve a single expert of a group of experts, the range of disagreement within a group of experts can reflect the level of uncertainty (EPA, 2009). A difficulty in using expert elicitation methods relates to translating the outcomes into probabilities. One approach is to use expert elicitation to provide parameter estimates for use in Monte Carlo analysis (EPA, 2009).

The estimation of probability distributions does not account for individual preferences or attitudes toward uncertainty (Heal et al., 2005) nor does it consider the (un)certainty of individuals about their own preferences regarding ecosystem services (Pascual et al., 2010). Scenario analysis is an additional approach to considering uncertainty in ecosystem valuation, whereby several scenarios are generated and compared based on different parameter estimates that represent various possible future scenarios (Turner et al., 2008).

A data enrichment or data fusion approach is proposed by Pascual et al. (2010) as a means of addressing technical and preference uncertainty, whereby different valuation methods (generally revealed and stated preference methods) are combined in estimating the value of an ecosystem service. A similar process is referred to by Heal et al. (2005:123) as data pooling and described as “taking data from different valuation methods and using the combined data, typically from two valuation methods, to estimate a single model of preferences”. While having not yet been extensively applied, the approach is seen as advantageous in that information about actual behaviour (revealed preferences) is combined with stated-preference data on hypothetical behaviour (where observations of actual behaviour are not possible), thus increasing the amount of information available as well as providing a means of cross-validating findings (Pascual et al., 2010 citing Haab and McConnell, 2002). As noted by Heal et al. (2005), the challenges of applying each of the methods remain and, further, the issue of how to weight the stated-preference and revealed preference data in the final model must be addressed.

In dealing with valuation under risk, the risk must be characterized to some extent and the decision-maker’s attitude towards risk should be known (Heal et al., 2005). Using an example of an aquifer and uncertainty regarding recharge rates for the aquifer, Heal et al. (2005) describe how marginal values can be influenced by the recognition of risk. The authors note that under conditions of uncertainty, a risk averse decision maker will attribute additional value to a slightly higher stock of a good or service (e.g. stock of water in the aquifer) given uncertainty regarding the future level of stock (e.g. aquifer recharge rate). In this case, the marginal value of a unit of the good or service will be higher because of the risk.

In reality, ecological variables are by nature stochastic, ecological processes are incompletely understood and non-linear relationships are likely, as such some level of uncertainty will always be present. The concepts of ‘safe minimum standards’ and the precautionary principle have been proposed in cases of decision-making under uncertainty (Turner et al., 1998; Perman et al., 2003; Heal et al., 2005).
5 Conclusion

Working with nature is viewed as the centre of the transition to a green economy and ‘investing in ecological infrastructure’ and ‘proactive investment in natural capital’ are fundamental building blocks of the transition. For such investments to be successful, decision-makers need instruments and measures to support decision-making and avoid inappropriate or unintended trade-offs. Assessing the benefits and losses from investment decisions across stakeholders, income groups, location and time is considered crucial to avoiding inappropriate trade-offs (ten Brink et al., 2012). Demonstrating the contribution of ecological infrastructure and the value (or loss) of ecosystem services through case study examples can contribute to a more comprehensive and transparent evidence base of trade-offs in decisions. Such an evidence base can lead to better, more cost-efficient decisions, avoid inappropriate trade-offs and refocus economic and financial incentives to align with sustainability goals (ten Brink et al., 2012). The intended use of the valuation information generated through this study is to contribute to such an evidence base.

The aim of the study is to provide examples and information to demonstrate the potential benefit to society and the green economy of investing in ecological infrastructure. A case study approach will be taken. The methods and tools of neoclassical welfare economics (based on the efficiency goal) will be applied, while acknowledging the additional social goals of fair distribution and sustainable scale. A key challenge or possible limitation to the study is the, often, onerous data requirements of economic valuation methods. The proposed study aims to address this challenge by working with a multidisciplinary team through a broader project where research regarding the ecological (hydrological), social and economic aspects will be undertaken concurrently.

While acknowledging other dimensions of value, the proposed study will consider the value of ecological infrastructure from an anthropocentric perspective; the contribution it makes to human well-being. The study takes an ecological economics perspective in viewing the economic system as embedded within the broader socio-ecological system and considers the well-being of humans and ecosystems as interdependent. The objective of the proposed study is to investigate the human well-being implications, as measured through economic value, of an investment action taken to change/improve the health of the ecological infrastructure of the uMngeni River catchment, Kwazulu-Natal, South Africa. The study is concerned with changes in the ecosystem(s) and the changes in human well-being that result from an investment in ecological infrastructure. The value concept of interest is the value of relatively small changes in ecosystem service quality and quantity (marginal value). Additional attention will be given to considering ecosystem valuation under risk and addressing non-linear relationships such as the case of ecological thresholds.

5.1 Case Study Options

Case studies will be selected in collaboration with the broader research team based on relevance and data availability. Case studies will be selected to cover a variety of ecosystem benefits and locations along the uMngeni River system with the aim of demonstrating the value of ecological infrastructure to human well-being. The following criteria will inform the selection:

- Data availability, planned data collection as part of the broader project and relevance to research purpose;
- Site location and ecosystem benefit diversity;
- Potential for investigating ecological threshold and uncertainty (risk) effects;
- Priority ecosystem services and benefits (based on existing demand and highlighted through project related ecosystem service assessment processes).
The following three options have been suggested by the project team and will be considered as case study options. The rehabilitation of a wetland downstream of the Mpophomeni community and upstream of the Midmar water supply reservoir. This site is located in the upper uMngeni catchment. The wetland rehabilitation is planned as part of the waste water treatment infrastructure for the Mpophomeni settlement. The rehabilitation of the Baynespruit tributary to the uMnsunduze River. This site is located in the uMnsunduze basin in an urban setting (Pietermaritzburg City). The aim of the rehabilitation is to improve the water quality of the Baynespruit stream. The water quality of this system impacts on farmers and residents in the Sobantu suburb of Pietermaritzburg. The Palmiet tributary to the uMngeni River. This site is located in the lower uMngeni catchment in an urban setting. The Palmiet drains a relatively small catchment northwest of Durban and is under pressure from industrial and residential development. The system provides benefits to the local industries and issues of environmental stability and stream management have been raised. The relationship between industries and the river system and questions of governance and institutional arrangements form part of the broader research project.

The focus of this report has been a broad introduction to the methodology and principles of economic valuation as applied in the context of valuing ecosystem services and benefits. The next step is to define specific questions that can be investigated through the uMngeni system case study options and provide examples of how investments in ecological infrastructure benefit society and contribute to a transition to a green economy. Such benefits could take several forms such as increased delivery of ecosystem services, using built and ecological infrastructure together to achieve public goals (water quality standards), developing ‘green’ businesses or niches and creating ‘green’ jobs etc. For example, through the Midmar wetland rehabilitation case study, the benefits of combining ‘grey’ and ‘green’ infrastructure in wastewater treatment could be considered. This would be an example of investing in ecological infrastructure to comply with legislation and regulation. In the context of the Baynespruit Stream rehabilitation, opportunities for sustainable agriculture ventures as a result of improved water quality for irrigation could be investigated. Historically, farmers of the Sobantu community used the stream for small-scale agriculture; with a decline in the water quality of the stream crop cultivation has declined due to health concerns. Water pollution from industries is a concern in both the Baynespruit and Palmiet systems. Investigating the options and costs involved for such industries to use alternatives for dealing with wastewater could be an additional research avenue and an opportunity for engaging industries rehabilitating these systems.

Ten Brink et al. (2012) suggest several examples of ‘value for money from natural capital’ and ‘saving money through natural capital investments’ as part of the transition to a green economy, for example:

- Natural solutions for water regulation, filtration and treatment can filter and clean much more cheaply than water treatment plants,
  - Businesses (beverage and water companies) have found it useful to compare the costs of having to invest in water filtration installations to ensure appropriate water quality;
- Ecosystems can help reduce the likelihood and scale of extreme events;
- There are often multiple benefits of ecosystem conservation and restoration beyond the original objective,
  - while conserving biodiversity, protected areas often have the added benefits from tourism of as a source of local earnings and employment;
  - natural water retention and regulation can be more cost-effective than engineered solutions in the long run and provide co-benefits such as carbon storage, landscape value, recreation, and biodiversity.

The uMngeni system case studies provide an opportunity to investigate and provide similar examples in the context of South Africa’s transition to a green economy.
6 References


Crookes, D.J., 2012. Modelling the ecological-economic impacts of restoring natural capital, with a special focus on water and agriculture, at eight sites in South Africa. Doctor of Philosophy Economics. Faculty of Economic and Management Sciences, Stellenbosch University.


Island Press, Washington, DC.


Rebelo, A.J. 2012. An ecological and hydrological evaluation of the effects of restoration on ecosystem services in the Kromme River System, South Africa. Master of Science. Faculty of Agriscience, Stellenbosch University.


7 Appendices
### Appendix A: Examples of valuation information use in influencing decision-making

<table>
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<tr>
<th>Use category</th>
<th>Use</th>
<th>Example</th>
<th>Reference</th>
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<tbody>
<tr>
<td>Decisive: trade-offs</td>
<td>Evaluate the environmental, social, and/or economic impact of a proposed development or policy</td>
<td>In California, a valuation influenced a decision by the regional water board to require the County of Los Angeles to divert storm water runoff to the local sewage treatment plant in order to improve coastal water quality. The study showed that the health benefits of reduced storm water flow far outweighed the cost of the diversions.</td>
<td>Given et al. (2006)</td>
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<td>Informative: justification &amp; support</td>
<td>Justify, support, inform, and/or advocate policies that protect or sustainably use (coastal) ecosystems</td>
<td>In St. Maarten, a valuation found that coral reefs inside a proposed marine park contributed $58 million per year to the local economy through tourism and fisheries. These findings helped convince the government to establish the Man of War Shoal Marine Park—the country’s first national park.</td>
<td>Bervoets (2010)</td>
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<td>Informative: awareness-raising</td>
<td>Raise awareness of the value of (coastal) ecosystems</td>
<td>A valuation showed that coral reefs and mangroves contribute to a significant portion of Belize’s GDP. These results supported action on multiple fronts, including a landmark Supreme Court ruling to fine a ship owner for a grounding on the Mesoamerican Reef, the government’s decision to enact a host of new fisheries regulations, and a successful civil society campaign against offshore oil drilling.</td>
<td>Cooper et al. (2009)</td>
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<tr>
<td>Informative: accounting indicators</td>
<td>Inform green national accounting</td>
<td>The governments of Namibia, Norway, Iceland, the Philippines, and the United States have created integrated environmental and economic accounts for marine fisheries. Tracking economic values associated with fish stocks and harvests over time helps fisheries managers and policy makers design policies for sustainable fisheries management.</td>
<td>FAO (2004)</td>
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<td>Technical: damage compensation</td>
<td>Establish levels of damage compensation</td>
<td>Valuation results were used to establish a schedule of escalating fines for injury to live coral in Florida, with assessed fines based on the area of impact. As a result, the Florida Keys National Marine Sanctuary has recovered millions of dollars for reef restoration after ship groundings.</td>
<td>Leeworthy (1991)</td>
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<tr>
<td>Technical: price-setting</td>
<td>Determine appropriate charging rates for environmental use (e.g., marine park user fees)</td>
<td>Several valuations justified the Bonaire Marine Park’s adoption, and later increase, of user fees, making it one of the few self-financed marine parks in the Caribbean.</td>
<td>Dixon et al. (1993), Uyarra (2002), Uyarra et al. (2010), Thur (2010)</td>
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<tr>
<td>Technical: price-setting</td>
<td>Design methods to extract finances from coastal ecosystem services (e.g., PES)</td>
<td>A valuation justified the establishment of a PES scheme in Honduras in which the tourism sector will pay a national park to maintain coastal water quality in collaboration with the palm oil industry.</td>
<td>PNUMA (2013)</td>
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<tr>
<td>Decisive: trade-offs</td>
<td>Compare costs and benefits of different uses of the coastal environment and assess trade-offs</td>
<td>A valuation played a key role in the development of Belize’s national Integrated Coastal Zone Management Plan (currently in draft form) by comparing ecosystem services provision and value under three coastal zoning scenarios developed iteratively with stakeholders: “conservation,” “development,” and “informed management.”</td>
<td>Clarke et al. (2013)</td>
</tr>
<tr>
<td>Decisive: trade-offs</td>
<td>Determine the most cost effective strategy for meeting a specific policy objective (e.g., coral reef health, water quality, climate change adaptation)</td>
<td>A valuation assessed 18 potential initiatives related to conservation, ecotourism, fisheries, and sustainable development in the Bahamas’ Exuma Cays. The study ranked the initiatives using criteria of costs, benefits, and feasibility. The study aims to influence land and sea use plans and the ongoing discussion about new regulations for the area.</td>
<td>Hargreaves-Allen (2012)</td>
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Source: Drawn from Waite et al. (2014) and related to ‘use categories’ proposed by Laurans et al. (2013), references from Waite et al. (2014).
## Appendix B: Ecosystem valuation frameworks and guidelines

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<thead>
<tr>
<th>Framework</th>
<th>Reference</th>
<th>Publication</th>
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<tbody>
<tr>
<td>An appraisal framework for wetland valuation</td>
<td>Barbier et al. (1997)</td>
<td>Ramsar Convention report</td>
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<tr>
<td>Phases of a joint ecosystem assessment and economic analysis for a single scenario</td>
<td>Bateman et al. (2011)</td>
<td>Environmental and Resource Economics article</td>
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<tr>
<td>Benefit valuation steps</td>
<td>Brouwer &amp; Georgiou (2012)</td>
<td>WHO report</td>
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<td>The 'Impact Pathway' approach to the valuation of ecosystem services</td>
<td>DEFRA (2011)</td>
<td>UK Government report</td>
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<td>An integrated and expanded approach to ecological valuation</td>
<td>EPA (2009)</td>
<td>USA Government report</td>
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<td>The economic approach to valuation</td>
<td>Heal et al. (2005)</td>
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<td>The ecosystem valuation framework</td>
<td>Hein et al. (2006)</td>
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<td>Template for the assessment and valuation of water quality-related services</td>
<td>Keeler et al. (2012)</td>
<td>PNAS article</td>
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<td>Practical steps in the execution of cost-benefit analysis</td>
<td>Mullins et al. (2014)</td>
<td>SA Water Research Commission report</td>
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<td>A stepwise approach to assessing nature’s benefits</td>
<td>TEEB (2010)</td>
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<td>A protocol for the quantification and valuation of wetland ecosystem services</td>
<td>Turpie &amp; Kleynhans (2010)</td>
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<td>The ecosystem services approach: Valuation of multi-functional wetlands</td>
<td>Turner et al. (2008)</td>
<td>Book, published by EarthScan Ecosystem Services article</td>
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<td>Methodological framework of Steart ecosystem services assessment study</td>
<td>Vieira et al. (2014)</td>
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<tr>
<td>Steps in conducting coastal ecosystem valuation to inform decision making in the Caribbean</td>
<td>Waite et al. (2014)</td>
<td>World Resources Institute report</td>
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Note: Highlighted references refer to South African publications

Source: Authors own review.
References to Appendix B: Ecosystem valuation frameworks and guidelines


