A FRAMEWORK FOR THE ECONOMIC VALUATION OF WETLAND REHABILITATION: CASE STUDIES FROM SOUTH AFRICA

A thesis submitted in fulfilment of the
requirements of the degree of

DOCTOR OF PHILOSOPHY
of
RHODES UNIVERSITY

by

MICHELLE BROWNE
ORCID ID
https://orcid.org/0000-0001-5286-8563

March 2022

ABSTRACT

Wetlands are recognised as having the potential to contribute long-term benefits to society; wetland rehabilitation is undertaken to recover these benefits in response to widespread wetland degradation. Increasingly, there have been calls to value the benefits of wetland rehabiliation to justify further investment. Such is the case in South Africa. Furthermore, recent global agendas and targets for ecosystem restoration, such as the declaration of the Decade of Restoration 2021-2030, suggest increasing pressure on governments to implement rehabilitation and imply a concomitant increase in decision-making regarding where and how to rehabilitate.

In response to these information needs, this thesis explores the economic valuation of wetland rehabilition through a narrative review of the foundational theory of values and valuation, a quantitative review of applied wetland rehabilitation economic valuation studies, and the evaluation of five wetland rehabilitation projects from South Africa. Projects were selected as case studies to represent various rehabilitation goals and explore different contexts (urban-rural; beneficiary groups), the timing of the evaluation (ex ante, ex post) and value types and valuation methods. The final chapter of the thesis integrates the case study experiences with the findings of the theoretical research components to propose a framework for the valuation of wetland rehabilitation, which can be applied in South Africa, and more generally, to further demonstrate the values of wetland rehabilitation, and as a tool to guide wetland rehabilitation decision-making.

While initially grounded in mainstream economics, the research led into a number of fields including philosophy, social-ecological systems and social-ecological relations thinking,

several environmental science areas and livelihood and human well-being frameworks. A deeper look into economic theory and history revealed an evolution of thinking on the meaning of ‘value’ and view of ‘nature’ and numerous critiques of standard neoclassical economics.

From the insights gained and the case study experiences, this thesis argues that the neoclassical economic perspective, especially combined with a monetary metric, is too restrictive, and arguably too abstract in its assumptions of human behaviour and reliance on mathematical models, as an overarching framework for the valuation of wetland rehabilition. This is not to suggest that standard economic valuation concepts and methods cannot be useful, as the research case studies illustrated, but rather that wetland valuation must be approached from a value pluralism perspective. To this end, the proposed framework offers a way to think beyond, or in addition to, standard economic approaches in articulating the values of wetland rehabilitation.

PREFACE AND ACKNOWLEDGEMENTS

This research has benefited from, and contributed to, two broader research projects both funded by the South African Water Research Commission (WRC).

The first project, WRC project K5/2344, related directly to wetland rehabilitation evaluation and the development of an integrated monitoring and evaluation framework to assess wetland rehabilitation in South Africa - WET-RehabEvaluate Version 2 (Walters et al., 2019). The framework encompasses a set of principles and criteria, a step-by-step guide, and a series of detailed modules for monitoring and evaluating specific aspects of wetland rehabilitation. This research contributed specifically to the evaluation of the Manalana, Xharas and Edendale Mall wetland rehabilitation projects and the development of the detailed module on the 'economic evaluation of the outcomes of wetland rehabilitation', which I led.

The second project, WRC Project K5/2354, had a broader focus and aimed to identify where and how investment in the protection and / or restoration of ecological infrastructure within the uMngeni River catchment of KwaZulu-Natal could be made to generate sustainable returns in terms of water security assurance. The research culminated in the creation of the Opportunities and Risks Framework for Investing in Ecological Infrastructure (ORF4Ei) which can be used as a framework to evaluate where, how and by whom, ecological infrastructure and other water security interventions can be made in the uMngeni River catchment (Jewitt et al., 2020), and 'Ten Lessons for Investment in Ecological Infrastructure'. This research contributed specifically to the evaluation of the Mthinzima wetland and Baynespruit Stream case studies, which I led.

The conception, design, and execution of this work are my own; the WRC project teams are duly acknowledged for their technical guidance and contributions made during the case study evaluations. My contribution to the WRC projects included conducting the economic valuation assessments, which involved the study design, data collection, data analysis and write-up of case study reports, and contribution to the overall synthesis of findings, framework development and writing of the final project reports.

I would like to express my very sincere thanks to the following people and organizations for their assistance in the pursuance of this research.

Prof. Gavin Fraser, Supervisor (Department of Economics and Economic History, Rhodes University)
Prof. Jen Snowball, Supervisor (Department of Economics and Economic History, Rhodes University)

All those involved in the WRC projects including the research teams, project reference groups, reviewers and WRC team, and particularly:

Ms Nonthando Buthelezi (University of KwaZulu-Natal)
Mr Craig Cowden (GroundTruth Water, Wetlands and Environmental Engineering)
Dr Stuart Ferrer (University of KwaZulu-Natal)
Ms Megan Grewcock (GroundTruth Water, Wetlands and Environmental Engineering)
Prof Graham Jewitt (University of KwaZulu-Natal; IHE Delft Institute for Water Education)
Dr Donovan Kotze (University of KwaZulu-Natal)
Mr Bonani Madikizela (Water Research Commission)
Mr Myles Mander (FutureWorks)
Mr Bigboy Mkhabela (Association for Water and Rural Development)
Dr Lutendo Mugwedi (Institute of Natural Resources)
Dr Catherine Sutherland (University of KwaZulu-Natal)
Dr Damian Walters (South African National Biodiversity Institute).

All those who participated in the interviews, workshops and focus groups, and particularly to:

The Craigieburn Community
Mr Ian Felton (KZN Department of Economic Development, Tourism & Environmental Affairs)
Mr Steve Terry (Umgeni Water).

GreenMatter [ZA] (and Canon Collins) for a bursary in support of the PhD research.

The Lloyds Register Foundation (University of the West of England - Advanced Academic and Professional Development Programme) for a bursary in support of the PhD research.

Finally, a word of major thanks, for bearing with me through this one, to all my colleagues, friends and family, foremost, of course, to Jono.

I acknowledge that all references are accurately recorded and that, unless otherwise stated, all work herein is my own. I certify that this thesis has not been submitted for a degree at any other university. However, the following articles, conference papers and final reports have appeared:

BROWNE, M., FRASER, G. AND SNOWBALL, J., 2018. Economic evaluation of wetland restoration: a systematic review of the literature. *Restoration Ecology*, 26 (6), pp. 1120-1126. (Research conceptualised, analysed and written up by M Browne, with advice and assistance from G Fraser and J Snowball in their supervisory capacity).

BROWNE, M., WALTERS, D. AND KOTZE, D., 2017. *An economic evaluation of the livelihood benefits of a rehabilitated wetland, Mpumalanga Province, South Africa*. Presented at the 7th World Conference on Ecological Restoration, August 28 to September 1, 2017. Foz do Iguassu, Brazil. (Conference paper based on the Manalana wetland case study evaluation, Water Research Commission Project K5/2344. The evaluation was undertaken by D Walters and M Browne with contributions from D Kotze, C Cowden, M Grewcock and B Makhabela. M Browne undertook the economic evalaution and prepared and presented the paper).

WALTERS, D., KOTZE, D., COWDEN, C., BROWNE, M., GREWCOCK, M., JANKS, M. AND EGGERS, F., 2019. WET-RehabEvaluate Version 2: *An integrated monitoring and evaluation framework to assess wetland rehabilitation in South Africa*. WRC Report No. 2344/1/19. Pretoria: Water Research Commission. (This report was the final deliverable of the Water Research Commission Project K5/2344. M Browne prepared section 5.3, the detailed module on the 'economic evaluation of the outcomes of wetland rehabilitation' with review by M Mander and D Kotze).

BROWNE, M., 2019. *Benefits and costs of investing in ecological infrastructure: An economic value perspective*. Presented at the 8th World Conference on Ecological Restoration, September 24-28, 2019. Cape Town, South Africa. (Conference paper based on the Mthinzima wetland case study evaluation, Water Research Commission Project K5/2345. The evaluation was undertaken by M Browne with contributions from N Buthelezi, S Ferrer and G Jewitt. M Browne prepared and presented the paper with review from G Jewitt).

JEWITT, G.P.W., SUTHERLAND, C., BROWNE, M., STUART-HILL, S., RISKO, S., MARTEL, P., TAYLOR, J. AND VARGHESE, M., 2020. *Enhancing water security through restoration and maintenance of ecological infrastructure: lessons from the uMngeni River Catchment, South Africa*. WRC Report No. TT 815/20. Pretoria, South Africa: Water Research Commission. (This report was the final deliverable of the Water Research Commission Project K5/2345. M Browne prepared Chapter 3 'Investing in ecological infrastructure – snapshots from an economic perspective' with inputs from G Jewitt and contributed to the development of the 'Opportunities and Risks Framework for Ecological Infrastructure in the uMngeni River' (Chapter 6) and the 'Ten Lessons for Investment in Ecological Infrastructure').

This thesis is dedicated to my father, my foundation.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1: INTRODUCTION

1.1 A BRIEF CONTEXT

Wetlands are recognised as having the potential to contribute long-term benefits to society and human well-being and the practice of restoring wetland ecosystems is undertaken to (partially) recover these benefits in response to widespread wetland degradation. Ecosystem restoration is gaining recognition as an important environmental management tool (Ehrlich and Pringle, 2008; Goldstein et al., 2008; Day et al., 2009; Rey Benayas et al., 2009; Aronson et al., 2010; Robbins and Daniels, 2012; Wortley et al., 2013; Blignaut et al., 2014) and is increasingly cited in environmental policy and management strategies (Suding, 2011; Wortley et al., 2013; Blignaut et al., 2014) and the objectives of international bodies (Rey Benayas et al., 2009; Aronson and Alexandra, 2013; de Groot et al., 2013). This is most recently evident in the declaration of 'The Decade on Ecosystem Restoration, 2021-2030' by the United Nations as one of their key Environment Programs (United Nations Environment Programme & Food and Agriculture Organization of the United Nations, 2019).

In South Africa, the degradation and loss of wetlands has been significant. Remaining wetlands make up only 2.4% of South Africa's landscape, and 48 % of wetland ecosystem types have been classified as critically endangered (Nel and Driver, 2012). It is estimated that South Africa has lost approximately 50% of the original wetland area (Department of Environment Forestry and Fisheries (DEFF), 2021). In response, extensive investment in wetland rehabilitation[1] has occurred (Phillips and Madlokazi, 2011; Kotze and Ellery, 2008), supported by South African national policy (Armstrong, 2008). The Working for Wetlands Programme, a government supported Expanded Public Works initiative, has invested over R1.3 billion[2] in the rehabilitation of more than 1 500 wetlands since 2004 (DEFF, 2021). Given the investment in wetland rehabilitation in South Africa and the often costly nature of ecosystem restoration it is important that the outcomes of such investments be evaluated

[1] While ecosystem rehabilitation and restoration have different goals and strategies, they share a similar focus (SER, 2004) and the terms are, at times, used interchangeably (Grenfell et al., 2007; Blignaut et al. 2010). In wetland practice in South Africa, the term 'rehabilitation' is used and encompasses the definition of (ecosystem) restoration, rehabilitation and repair as defined by McDonald et al. (2016); however, a system that is rehabilitated is not expected to be restored back to its reference state/benchmark (section 2.2.3.2).

[2] 'R' refers to the South African Rand (ZAR). As at 28/09/21, US Dollar = R15.05 and Euro = R17.58.

(Kotze and Ellery, 2008). Yet, comprehensive evaluations of wetland rehabilitation projects in South Africa are lacking (van Zyl et al., 2004; Cowden *et al*., 2013; McConnachie et al., 2013).

Globally, there is a general absence in evidence-based evaluations of restoration outcomes (Palmer and Filoso, 2009; Caffey et al., 2014; Meli et al., 2014) or a lack in the transfer of this knowledge to the public (Palmer et al., 2007). While the benefits of restoring ecosystems are frequently cited, their quantification and valuation is less common (van Zyl et al., 2004; Rey-Benayas et al., 2009; Blignaut et al., 2013; Wortley et al., 2013). To justify increasing investment (often publicly funded) in ecosystem restoration, there is a growing need to provide evidence of the outcomes of restoration (Ntshotsho et al., 2011; Blignaut et al., 2013), identify and value the benefits to society of restoration activities (Weber and Stewart, 2009; Aronson et al., 2010; Pendleton and Baldera, 2010; Blignaut et al., 2013), and define the conditions and institutional arrangements under which the greatest benefits can be achieved (Goldstein et al., 2008; Suding, 2011).

In a review of the literature, Wortley et al. (2013) reported that only 3.9 percent of restoration evaluations considered the economic outcomes of restoration. Similarly, Aronson et al. (2010) found socioeconomic attributes of restoration to be underrepresented in the literature and Blignaut et al. (2013) noted that policy outcomes are rarely the focus of restoration research. Figueroa (2007) concluded that the economics of rehabilitating natural ecosystems 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).

1.2 GOALS OF THE RESEARCH

The intention of the research is to explore the economic valuation of wetland rehabilitation with the aims of quantifying and valuing the outcomes of a sample of wetland rehabilitation projects in South Africa (aim 1) and providing an approach for generating estimates of the value of wetland rehabilitation for use in South Africa (aim 2).

The research is intended to contribute to a broader undertaking to develop an integrated monitoring and evaluation framework to assess wetland rehabilitation in South Africa, funded by the Water Research Commission[3]. The broader project identified a need to incorporate both social and ecological aspects of wetland rehabilitation into a monitoring and evaluation framework recognising that a wetland is a complex system interlinked with, and influenced by, social factors. It is envisioned that the research will contribute to the theory and practice of valuing the outcomes of wetland rehabilitation and add to the body of knowledge of ecological restoration economics in South Africa.

1.3 THE METHODOLOGICAL APPROACH AND THESIS STRUCTURE

Underlying all research are philosophical presuppositions about what constitutes reality and 'valid' knowledge, how knowledge can be acquired, and which methods are appropriate. There are multiple ways of categorising theoretical perspectives on the research process. Within the social sciences, two common groupings are positivism, which is recognized as having had a dominant role within the social sciences, and post-positivism. Other theoretical perspectives within the social sciences include the interpretive, constructivist and critical perspectives (constructivist perspectives are sometimes grouped with interpretivist)[4].

A common distinction between positivism and alternative research perspectives is a move away from a commitment to knowledge as objective. Post-positivism views knowledge as subjective, influenced by social and cultural perspectives (McGregor and Murnane, 2010). A further difference is a shift from the focus of positivism on experimental and quantitative methods to uncover ‘facts’ to qualitative and inductive methods to gather broader information and interpret, reveal and explain. Particularly within an interpretivist perspective, the purpose of research is viewed as being ‘to interpret’, ‘to reflect understanding’; there are ways of knowing aside from the scientific method and understanding is contextual (Willis, 2007).

Post-positivist research approaches have been suggested as more suitable in addressing complexity (Wildemuth, 1993; Norton and Noonan, 2007). Ecosystems and environmental challenges are characterized by complexity (Söderbaum, 1990). As such, for this study, a post-

[3] Details provided in the acknowledgments.

[4] These perspectives are regarded by some as distinct paradigms, by others, as different epistemologies within a post-positivist paradigm (c.f., McGregor and Murnane, 2010).

positivist approach, which encompasses an interpretative methodology with the intent of understanding (Fischer, 1998; McGregor and Murnane, 2010), is considered most appropriate.

However, there are tensions between the philosophical views of neoclassical (standard or mainstream) economics and those developing within ecology and environmental management. The development of neoclassical economics was strongly influenced by the physical sciences and the associated assumptions of the positivist paradigm and is traditionally associated with a reductionist perspective and epistemological monism (Fullbrook, 2008; Hoover, 2008). In contrast, ecosystems are viewed from a holism perspective and regarded as having emergent properties which are a characteristic of complex systems (Cilliars, 2000). A holistic approach is increasingly considered appropriate to natural resource and environmental issues which are inherently complex (Prato, 1998). Furthermore, epistemological pluralism has been advocated in the context of linked social and environmental issues (Miller et al., 2008). While neoclassical economics is associated with monism at a theoretical level, Sent (2006) argued that applied, and increasingly theoretical, economics are diverging in their commitment to monism and applied mainstream economics may be better characterized as ‘moderate pluralism’.

Adopting an epistemological pluralist approach in research requires collaboration between disciplinary researchers and working out ways to accommodate different disciplinary views (Miller et al., 2008). Interdisciplinary and transdisciplinary approaches have been called for in studying complex social-ecological systems (Schoon and van der Leeuw, 2015; Esler et al., 2016) and, more broadly, for confronting ‘wicked problems’ (Max-Neef, 2005; Bhaskar et al., 2010; Vogel et al., 2016). It has been argued that ‘solutions’ to societal problems cannot be drawn from a single knowledge base (Hoffmann-Riem et al., 2008; Binder et al., 2013; Esler et al., 2016; Vogel et al., 2016) nor from a collection of independent disciplinary contributions (e.g., multidisciplinary) (Max-Neef, 2005; Baumgartner et al., 2008; Schoon and van der Leeuw, 2015). Broadly, interdisciplinarity can be understood as the integration of disciplinary knowledge involving co-operation between disciplines, and transdisciplinarity as some level of interaction and knowledge integration between academia and practice and/or society (Baumgartner et al., 2008)[5].

'Knowledge integration' is a central feature of inter- and transdisciplinary approaches. Bhaskar et al. (2010) argued that a lack of integration of knowledge – particularly in the fields of planning, economics and the policy-civil society interface – has undermined efforts to address, or even led to, complex societal problems such as climate change, biodiversity loss and resource degradation. An interdisciplinary approach is considered implicit in an ecosystem services framing and therefore in the economic valuation of wetland services, benefits and values (Turner et al., 2008). However, challenges to such collaborative efforts have been recognized, including differences between disciplines regarding time and spatial scales of analysis, forms of knowledge (e.g., qualitative vs quantitative), and levels of precision, accuracy and acceptance of uncertainty (Benda et al., 2002). As such, an interdisciplinary research process requires time for discussion and deliberation among disciplinary researchers.

The value of wetland rehabilitation necessarily involves the intersection of the social and the ecological. There are several theories describing the complex and interconnected relationship between humans and non-human nature. In the environmental valuation field, two theories have gained ground: social-ecological systems theory (e.g., Anderies et al., 2004; Gunderson et al., 2006; Ostrom, 2009), and social-ecological relations theory (e.g., Chan et al., 2016; Muradian and Pascual, 2018). Rather than being alternatives, these theories emphasize different elements of the interactions between humans and non-human nature and there are commonalities between them, and, to a degree, a complementarity (Binder et al., 2013). Shared across the concepts is an acknowledgement that humanity and non-human nature are not separable, but are interlinked, and influence and shape each other. Humans are a part of, not apart from, non-human nature; the delineation between 'social' and 'ecological' is artificial.

[5] These 'knowledge needs' are in addition to, not a replacement for, disciplinary knowledge. Disciplinary depth of knowledge is needed; how it is connected and assembled to address problems and effect change is considered critical (Bhaskar et al. 2010; Esler et al., 2016).

Social-ecological systems theory emphasises that a system is complex and adaptive, constituted of several interacting elements, and has the properties of non-linearity, uncertainty, emergence and self-organisation (Cilliars, 2000; Berkes, 2004). Systems thinking, complexity theory and adaptive management are integral elements of social-ecological systems theory. Wetlands are typically viewed as social-ecological systems (e.g., Olsson et al., 2004; Gunderson and Light, 2006), meaning that any evaluation of wetland rehabilitation must consider ecological variables (e.g., wetland characteristics), human variables (e.g., socioeconomic processes, structures of governance) and the interactions and feedbacks between these elements and how they influence one another.

From a social-ecological relations perspective, 'nature' or 'landscape' is viewed as the space where interactions among the ecological, social and cultural take place in a relational way (Muraca, 2011; Chan et al., 2016). A 'landscape', rather than being a fixed physical property, is constituted through its relations to other things; all elements, both biophysical and human, come together through their relations to produce an assemblage - the 'landscape' (Bassett and Peimer, 2015; Stenseke, 2018). In this way, landscapes are dynamic, spatially and historically contingent, and subjective, viewed differently by different people. This view emphasises the interdependent and dynamic ways in which the biophysical and social interact (Stone-Jovicich, 2015) and the diversity in how people relate to the environment. For example, 'stewardship' 'utilization' and 'devotion' can be viewed as different relational models, or ways humans connect or relate to elements of non-human nature (Muradian and Pascual, 2018). Both theories will be considered in shedding light on an approach for the valuation of wetland rehabilitation outcomes.

Broadly, the research takes an exploratory research approach integrating the review of several bodies of relevant literature (Chapters 2 and 3), with case study applications (Chapters 4, 5, and 6). The quantitative review approach of Pickering and Byrne (2014) is applied to review the published literature on the economic valuation of wetland rehabilitation to map the breadth and scope of existing literature on the topic (Chapter 3). The case studies explore the application of economic valuation in the context of the evaluation of the outcomes of wetland rehabilitation in South Africa.

Wetland rehabilitation interventions were chosen as case studies to represent different rehabilitation goals and priority outcomes including the goal of enhancing a particular ecosystem service or providing a specifc benefit (Manalana case study - livelihood benefits, Mthinzima case study – water quality enhancement, Baynespruit case study – address water security challenges); the goal of achieving an overall improvement across a range of ecosystem services (Xharas case study); and the goal of no nett loss of wetland habitat associated with development compliance regquirements (Edendale Mall case study). Furthermore, cases were selected to explore different contexts (urban-rural; beneficiary group), the timing of the evaluation (ex ante; ex post) and value types and valuation methods.

The case studies include two cases where economic valuation methods using a monetary metric are applied to measure the values associated with the priority rehabilitation outcomes, which are compared against the rehabilitation costs in a partial Cost-Benefit Analysis (CBA) (Chapters 4 and 5). A monetary metric of value was adopted largely in response to a call from management and implementation stakeholders for the research to provide conventional cost-benefit analyses of rehabilitation projects. The Mthinizima wetland case was chosen particularly because it demonstrated that, even though the replacement cost method is considered a conceptually simpler approach and commonly applied in ecosystem valuation, the method can still be challenging to apply and relies on several assumptions which may affect the reliability of the results. From a reflection on the case study application, a set of eight factors to consider when using the replacement cost method in wetland rehabilitation valuation and decision-making have been derived. The Manalana wetland evaluation applied a market-based monetary valuation with CBA, similarly considered a fairly straightforward approach, yet the case study illustrated that the simple cost-benefit decision making rule should not be applied without careful consideration of the evaluation context and emphasised the importance of contextualising monetary value estimates with respect to the specific situation of the beneficiaries. The study further demonstrated how these types of assessment leave out other, potentially critical, values, such as the safety net function of wetland resources as in the Manalana case.

Chapter 6 presents key insights from three additional evaluation case studies where monetary-based valuation was considered, but found to be not possible or appropriate, and alternative approaches are applied to explore other options for articulating the contribution and value of wetland rehabilitation. Chapter 7 synthesises the research findings into a guiding framework for the valuation of wetland rehabilitation and concludes the thesis.

CHAPTER 2: FOUNDATIONS: VALUES, WETLANDS AND VALUATION

2.1 ENVIRONMENTAL VALUES

"Values are not fragile or rare or delicate or endangered. We do not live in an axiological desert but in a rain forest. Everywhere the air is thick with them" (Weston, 2009:1).

Ecosystem restoration is regarded as an elective action, implying a choice in respect of whether to invest or not (Clewell and Aronson, 2006). Any decision between competing alternatives implies some form of valuation, implicit or explicit (Costanza, 2000). In restoring ecological systems, an individual or society perceives itself as restoring something that matters, something of value. However, ecosystems and their attributes matter to people in different ways and at different times or in different contexts. For O'Neill et al. (2008) humans 'live from', 'live in' and 'live with' non-human nature; these various kinds of human-nature relations give rise to different sources and types of value.

Perceptions of, and normative stances on, environmental values and how they are generated and held are diverse (Bockstael et al., 2000; de Groot et al., 2006; EPA, 2009) and the field of environmental values is itself "active, multifaceted and disputatious" (Satterfield and Kalof, 2005:xxi). This section seeks to shed light on various understandings and conceptualisations of 'value' in the context of attributing value to nature and ecological restoration. There is an extensive body of literature across multiple disciplines which deals with the concept of environmental value (Dietz et al., 2005), an account of which is beyond the scope of this discussion. This section focuses on notions of value from the disciplines of philosophy and economics and introduces some additional perspectives from the ecosystem valuation field, and is by no means representative of the many theories of environmental value, but rather serves to emphasise that "value is not a single, simple concept" (EPA, 2009:13).

The first part of this section presents several philosophic concepts of value, particularly relevant, and much debated, within environmental philosophy and ethics. The second part focuses in on economic concepts of value specifically in the context of environmental values. The next sub-section touches on additional concepts from the ecosystem valuation field, and the section closes with a discussion of the multiplicity of values and the implication for the economic valuation of wetland restoration.

2.1.1 Philosophic concepts of value

The topic of ‘environmental value’ is perplexing, in part because the term ‘value’ is used in a number of very different ways; the Concise Oxford English Dictionary alone identifies six meanings. An initial distinction can be drawn between held values (beliefs or principles we adhere to) and assigned values (expressions of preference in the form of ranking, index or numeric metrics) (Satterfield and Kalof, 2005; Kenter et al., 2015). Environmental philosophy and ethics tend to focus on the broader concept of ‘held values’ addressing questions of what objects / phenomena hold value (e.g., anthropocentric vs non-anthropocentric views), whether there are multiple ‘independent’ values (value pluralism) or one ultimate single value (value monism) and whether phenomena hold value in their ‘own right’ independent of human goals (intrinsic value).

2.1.1.1 Value pluralism

Central to the debate on environmental values is the question of whether there is one ‘ultimate value’ to which all other values are reducible – value monism - or several distinct values – value pluralism - which are not reducible to a single ultimate value or to each other (O’Neill et al., 2008). A monistic approach is an attempt to represent all value in a single framework of analysis such as economic CBA or rights theory, whereas a pluralistic perspective does not impose “a universal vocabulary upon the discourse of environmental value” (Norton and Noonan, 2007:666). While value monism is attractive in that it appears to offer the possibility of a common (cardinal, or at least, ordinal) measure of value, thereby reducing decision-making to a matter of calculus, “a proper appreciation of the complexity of values, rather than an attempt to simplify them” has been advocated (O’Neill et al., 2008:87).

Light (2002:434) reasoned that “the sources of value in nature are too diverse to account for in any single theory” and that “the multitude of contexts in which we find ourselves in different kinds of ethical relationships with both humans and nature demand a plurality of approaches for fulfilling our moral obligations”. Similarly, Norton and Noonan (2007) argued that while a monistic value approach is taken as the ‘default’ for evaluating environmental and ecological change; the approach is ill-suited to addressing environmental issues which are 'Wicked Problems'[6]. While rejecting value monism, Norton and Noonan (2007) did not oppose economic valuation or consider an economic framing incompatible with value pluralism. They were concerned, however, with what they described as an enthusiasm for the ecosystem services concept used in conjunction with economic valuation methods, which has "locked the rhetoric of environmental evaluation in a very monistic, utilitarian, and economic vernacular that leaves little or no room for other social scientific methods, or for appeal to philosophical reasons or theological ideals" (Norton and Noonan, 2007:665).

'Value pluralism' is variously interpreted and is used to refer to value theory in the sense of the existence (or not) of an ultimate source of value (e.g., utility), this is the question usually tackled in philosophy. A second interpretation, and the one more commonly used in the environmental valuation field, is that of multiple types of value (e.g., economic, cultural, relational), which can all fall within one value theory (e.g., human well-being), but are not necessarily substitutable or comparable in terms of being reduced to a single 'measure' or 'expression'. In this sense, Lockwood (1997:83) identified "human values for natural areas", that is, the values humans hold for the environment, which may be of different types, but fall within a specific theory of value based on human preferences. In his framework, Lockwood (1997) suggested that human value expressions can be weakly comparable (comparable, but not completely 'rankable') and strongly comparable (completely 'rankable'). Strongly comparable preferences can be non-compensatory (comparable, but not substitutable[7]) or exchangeable (can be assessed against a common value standard).

2.1.1.2 Intrinsic and instrumental value

In environmental ethics, the notion of 'intrinsic value' and its application to nature is a fundamental question (Morito, 2003). Intrinsic value is present when "the referent entity is an end in itself, such that the value is autonomous and independent of any other entity" (Lockwood, 1999: 382); that is, the entity has value 'in its own right' in contrast to deriving its

[6] See Rittel and Webber, 1973:160-167 for a discussion of the distinguishing properties of wicked problems.
[7] That is, "a reduction in quantity or quality of the valued entity cannot be compensated for by a change in another entity" (Lockwood, 1997:85). This interpretation links with the concept of value incommensurability, a core principle of ecological economics (Martinez-Alier et al., 1998), see section 2.1.3.4.

value from some other entity or its contribution to achieving some other end (Zimmerman, 2015). In contrast, extrinsic value is present when the value of an entity derives from, or is defined in relation to, some other value (Baum, 2012). Intrinsic value is commonly contrasted with 'instrumental value'.

Instrumental value refers to the value something has as a means or instrument of attaining a particular end other than itself (Díaz et al., 2014; Spangenberg and Settele, 2016). It is the value derived from the object's usefulness in achieving a goal, which is in contrast to intrinsic value, which is value that exists independently of any such contribution or usefulness (Heal et al., 2005). Objects, activities and states of affair (e.g., ecosystems) have instrumental value insofar as they are a means to some other end, such as human welfare. As such, instrumental values vary with the 'ends' (goal or purpose) to which they contribute and, therefore, the context in which the valuation takes place; instrumental values are not absolutes (Spangenberg and Settele, 2016).

For example, the value of a wetland in providing crops (food) for human consumption is an instrumental value. The value of the wetland in providing a 'sink' for water pollutants and improving water quality for downstream use is also an instrumental value. However, these 'uses' are not necessarily mutually compatible and, further, the 'value' of these two benefits may be very different. The instrumental value of a specific wetland will depend on the local context (e.g., the preferences, or demand, for these alternative 'uses'). An intrinsic value of the wetland would reflect the value the wetland 'holds' that is independent of whether humans have any preference for, or derive and benefit from, the wetland.

Implicitly tied to the notion of instrumental value is 'substitutability', that is, entities bearing instrumental value may be replaced or compensated for with entities providing the same contribution to the 'end'. As explained by Muraca (2011:388): "*x* is instrumentally valuable when it yields utility as a means to *A*'s ends: it is not *x* that matters to *A*, but the benefit obtained by means of *x*. Any other entity that can yield the same utility is as good and valuable as *x*".

Some theorists propose that assigning intrinsic value is a way of making sense of the world in terms of helping people to understand how they do, and why they should, care about the environment (e.g., McShane, 2007). Others argue to discard the notion of intrinsic value within the environmental values and policy arena (e.g., Weston, 1985; Light, 2002; Morito, 2003). Arguments against intrinsic value tend to centre around two aspects: whether the concept is useful in addressing environmental concerns and doubts about the meaning and existence of intrinsic value. The concept of intrinsic value is considered, by some, as unhelpful, both in terms of being an ineffective motivator of environmental protection (Weston, 1985; Light, 2002), as well as being difficult to assess or measure (Dietz et al., 2005). On the second aspect, it is important to note that the 'intrinsic value' concept is variously interpreted. For example, in some instances it is taken to be any non-instrumental value; for others it derives from an inherent property an object possesses independent of human judgement (i.e., an objective value); it is also viewed as a reflection of the moral standing an entity possesses (Weston, 1985; O'Neill et al., 2008; Díaz et al., 2015, Zimmerman, 2015).

An interrogation of the philosophical debate on intrinsic value is beyond the scope of this thesis, however several points relevant to this study stand out: (i) the debate signals a sense by many of the importance or meaning of the natural environment beyond a narrow interpretation of instrumental value; (ii) it highlights the difficulties in attempting to define and assess or measure intrinsic value; and (iii) it brings to attention alternative conceptualisations of value systems such as the interdependent and relational characteristics emphasised by Weston (1985) and O'Neill et al. (2008), the notion of values as processes rather than as properties (Morito, 2003) and the importance of context in considering environmental values (Weston, 1985; O'Neill et al., 2008; Spangenberg and Settele, 2016).

2.1.1.3 Anthropocentric and non-anthropocentric value

Anthropocentricism is the view that humans are the base by which all morals are evaluated; the morality of an action is judged by its benefit to humans and improving human well-being. In the context of environmental values, anthropocentrism places human well-being at the core of the value attributed to nature; the value of non-human entities originates in their contribution to human well-being and the degree to which they serve human interests (Norton, 1984; Goulder and Kennedy, 1997). Within an anthropocentric value theory, intrinsic value is attributed only to human-beings whereas a non-anthropocentric view extends moral consideration and intrinsic value to non-human entities (Callicott, 1984; O'Neill et al., 2008). The utilitarianism ethical doctrine as applied in neoclassical economics falls within an anthropocentric value theory. Under utilitarianism, decisions are judged on their consequences where the best action is the one which produces the greatest good for the most people (Driver, 2014). In the neoclassical economic sense, the 'greatest good' is measured as the satisfaction of human preferences (utility) (Hodgson, 1993).

There is much debate over anthropocentrism as the fundamental principle of an environmental ethic and defining a non-anthropocentric perspective has been a central focus of environmental ethics (Grey, 1993; Weston, 2009); yet a widely accepted non-anthropocentric view has yet to emerge. Weston (2009) argued that given the thoroughly 'athropocentrized' nature of the Western world view, a non-anthropocentric ethic is, as yet, barely conceivable, while Grey (1993) and Hayward (1997) argued that within any theory or value type proposed by humans some degree of anthropocentrism is inevitable. In Hayward's (1997) words:

> "As long as the valuer is a human, the very selection of criteria of value will be limited by this fact. It is this fact which precludes the possibility of a radically nonanthropocentric value scheme, if by that is meant the adoption of a set of values which are supposed to be completely unrelated to any existing human values […] values are always the values of the valuer: so as long as the class of valuers includes human beings, human values are ineliminable" (Hayward, 1997:56-57).

Several points stand out from the anthropocentric – non-anthropocentric environmental value debate. Issues of the abstract nature of the concept of non-anthropocentrism for practice (Grey, 1993) and a general ambiguity and misinterpretation of anthropocentrism have been raised (Norton, 1984, Hayward, 1997, Light, 2002). Several theorists attribute much of the misunderstanding of anthropocentrism to a lack of attention to defining what is meant by human interests (Norton 1984; Grey, 1993). The importance of context (current and historical) in questions of environmental value has been emphasised by several scholars (O'Neill et al., 2008; Weston, 2009; Katz, 2011). Support for value pluralism (O'Neill et al., 2008; Weston, 2009; Katz, 2011) and the view that environmental values are complex

(Rolston, 1983; O'Neill et al., 2008) is evident across several arguments, from both supporters and critics of anthropocentrism. The significance of interactions or relations as a source of environmental value has been proposed (O'Neill et al., 2008; Weston, 2009) and the view that values are socially and culturally shaped has been argued (Weston, 2009; Díaz et al., 2014). A distinction between artificial and natural systems and the values they hold has been ventured (Katz, 1999) which implies a relationship between the value of an ecosystem and its origin and evolution and suggests value heterogeneity between the value of 'natural' and 'restored' ecosystems.

Weston (2009), in rejecting the anthropocentric - non-anthropocentric conceptualization of values, argued that values need to be situated in their contexts and that it is an understanding of the dynamics of values and ways of making values more apparent and discernible in every day experiences that should be pursued.

2.1.1.4 The value of 'restored' nature

Nature 'restoration' is considered a relatively recent paradigm of nature protection in contrast to nature 'preservation' which was the dominant paradigm of the 20th century. Preservationists locate nature's value in the degree to which it is independent of human influence, in its wildness or naturalness (Hettinger, 2012). The perspective conveys a sense of a world apart from humans, a separation of nature and humanity. Given ongoing environmental degradation and growing awareness of the influence of human activities on ecosystems, the attempt to restore degraded nature – nature restoration - has become a prominent environmental goal (Hettinger, 2012). Nature's value, from a restoration perspective, lies in its 'thriving biodiversity', rather than in a lack of humanization. For restorationists, humans are considered part of nature and the nature restoration paradigm promotes a positive vision of humanities' participation in nature (Hettinger, 2012).

The value of 'restored nature' depends on what is taken as the source of nature's value. Several views are illustrative. From one perspective, advocated for example by Robert Elliot (Elliot, 1982), origin (the process of creation) and history are important elements that constitute the value of nature; nature's value lies in its "causal continuity with the past" (Elliot, 1982:87). This means that specific landscapes and ecosystems become irreplaceable by virtue of having their own specific 'genesis' and history and therefore have no substitutes (O'Neill et al., 2008). The implication for ecological restoration is that a restored system would not hold the same value as the natural (pre-disturbed) system given the value imparted by a system's origin and history separately from its physical components. For Elliot (1982), ecological restoration cannot fully recreate the value that is lost when nature is disrupted. Elliot's views rests on treating 'nature' as unmodified by human activity and ecological restoration as an attempt to replicate 'nature'.

Like Elliot (1982), Katz (1992) argued that ecological restoration cannot recapture the value of nature. For Katz (1992), the 'naturalness' of a landscape is a determinant of its value, once disturbed through human actions the 'naturalness' is lost and even successful attempts to 'restore' the ecosystem functioning and flora and fauna of the original landscape cannot recapture the 'naturalness'. This view rests heavily on a particular interpretation of 'nature'[8]; that of a system / landscape absent from human agency. This view raises an immediate challenge to ecological restoration – termed the paradox of nature protection by O'Neill et al. (2008): how can something (nature) defined by the property of its independence or separation from humanity be 'restored' or 'protected' through human intervention? For Katz (1992; 2012), 'restored nature' is impossible; rather, a restored landscape is an 'artefact', created to meet human interests and invariably anthropocentric. So then, the value of 'restored nature' cannot be equivalent to the value of nature.

In the context of ecological restoration, Attfield (1994) proposed a view of nature's value less reliant on the origin and history of the system and its independence from human agency. For Attfield (1994:53), nature's value stems from a cluster of sources including: "the intrinsic value of the flourishing of the creatures which originate there, the value of intact ecosystems, and the value of the human appreciation of wildness, of otherness, and of living systems which originate from evolution alone and lack any human modification". While restored nature lacks the value associated with evolution, the flourishing of natural creatures - the

[8] The interpretation of 'nature' and what constitutes 'natural' is a much debated point within environmental philosophy and ethics and beyond the scope of this discussion. O'Neill et al. (2008) elaborate on the debate.

principal value of nature according to Attfield (1994) - occurs wherever ‘healthy’ ecosystems and biodiversity occur (Attfield, 1994).

The views presented thus far consider restoration, and the value thereof, in the sense of recovering a perceived natural or historical condition (although Attfield’s view begins to diverge). Conventionally, ecological restoration is associated with attempts to return an ecosystem to (some) historical state prior to anthropogenic disturbance (Higgs et al., 2014). However, establishing the natural or appropriate historical state of an ecosystem is complicated by the debate on what constitutes ‘natural’, the impacts of wide-ranging anthropogenic influences (e.g., climate change), the fact that ecosystems are subject to non-human driven variability and are themselves dynamic (especially over long timeframes), and knowledge gaps.

Further, with the emergence of the ‘natural capital’ and ‘ecosystem services’ framings of nature, greater emphasis has been placed on ecosystems to generate important ecosystem services (e.g., water quality enhancement by wetlands) and ecosystem restoration is increasing viewed as a means to achieve goals other than historical fidelity. Such goals, specifically those related to the provision of ecosystem services (desired functions), emphasise the instrumental value of ecosystems, and therefore, the instrumental value of ecosystem restoration. Rohwer and Marris (2016) contend that while the values of nature related to ‘origin and history’ and ‘naturalness’ are not restored through ecological restoration, other values can be. The authors go further, claiming that: “it is not impossible to imagine a restoration that results in a site with higher biodiversity, resilience, and community connection than any prior state, arguably making up quantitatively in these values what is unachievable in the value of naturalness” Rohwer and Marris (2016:678).

The various views on the value of restored nature express different ways in which human’s value things and different conceptions of nature, illustrating a range of potential values (and/or sources of value) associated with nature and ecological restoration. O’Neil et al. (2008) emphasized that ideas about ‘what is natural’ are affected by the cultural circumstances within which they are formed. Similarly, for Higgs (1997:347) “nature and ecosystems are historically and culturally contingent ideas”. A particular view of nature is based on a specific point in time and the cultural and social circumstances within which it is formed. The value of 'restored' nature depends on the valuer's conception of nature and its source(s) of value, and so too is historically and culturally contingent.

The economic value of restored nature is an expression of a particular type of value - instrumental value – associated with the benefits to humans that result from the restoration. In application, an economic framing reflects the value of independent outcomes associated with changes in the structure and function of an ecosystem as a result of restoration activities, rather than signifying the value of the restored ecosystem itself. The next section focuses the discussion of environmental values on the economic concept of value.

2.1.2 Economic concepts of value

Economic value is a specific type of value based on the goals of economics, a particular world view and various assumptions about the world and human behaviour. In the view of Bockstael et al. (2000:1386) "when the stated objective is to measure economic value, the concepts and principles associated with performing the task are not matters of opinion". To appreciate the meaning of the economic value of ecosystems and their attributes it is necessary to consider the goals and world view of economics and the principles and assumptions that underpin economic value.

This section opens by considering how economics as a field is defined, the overarching economic world view (ontology) and the changing view of nature in economics over time. Following these introductory sub-sections, various theories and concepts underpinning the neoclassical economics interpretation of value are discussed. An economic value typology is presented in the final sub-section.

2.1.2.1 Economics – variously defined

In a general sense, the English term 'economics' derives from the Greek word 'oikonomia' integrating 'oikos' translated as 'household' and 'nemein' translated as 'management and dispensation' (Leshem, 2016). As such, 'economics' is loosely translated from ancient Greek to mean 'management of the household'. Leshem (2016:226) drew parallels between the ancient Greek meaning of 'oikonomia' and contemporary economics by suggesting that both deal with the "study of human behavior as a relationship between ends and means which have alternative uses", but argues that the two differ considerably in their relationship to ethics. Leshem (2016) suggested that 'oikonomia' was deeply rooted in ethical judgements, while contemporary economics is, in principle, independent of any ethical position.

Over time, the meaning of 'economics' has evolved with three notable definitions being distinguished, that of the economists Adam Smith (1723-1790), Alfred Marshall (1842-1942) and Lionel Robbins (1898-1984). The following discussion briefly outlines this evolution, for a more comprehensive treatment see Backhouse and Medena (2009). According to Smith (1776:557):

> "Political economy, considered as a branch of the science of a statesman or legislator, proposes two distinct objects: first, to provide a plentiful revenue or subsistence for the people, or more properly to enable them to provide such a revenue or subsistence for themselves; and secondly, to supply the state or commonwealth with a revenue sufficient for the public services. It proposes to enrich both the people and the sovereign".

Smith's definition was refined in the early 19th century to a science that concerns the production, distribution, and consumption of wealth and become known as the 'wealth-based' definition of economics and dominated during the 19th century (Backhouse and Medena, 2009).

Towards the end of the 19th century, the focus of economics expanded from wealth accumulation to the study of human behaviour, leading to Alfred Marshall's definition of economics. For Marshall (1920: I, I):

> "Political economy or economics is a study of mankind in the ordinary business of life; it examines that part of individual and social action which is most closely connected with the attainment and with the use of the material requisites of wellbeing. Thus it is on the one side a study of wealth; and on the other, and more important side, a part of the study of man".

To Marshall, wealth consisted of desirable things "things that satisfy human wants… but not all things that are desirable are reckoned as wealth. The affection of friends, for instance, is an important element of wellbeing, but it is not reckoned as wealth" (Marshall, 1920: II, II).

Marshall goes on to enumerate the main questions economic science should address, including "[s]ubject to what limitations is the price of anything a measure of its desirability?" (Marshall, 1920:I, IV).

Lionel Robbins proposed the still contemporary definition of economics, introducing 'scarcity' as a central element: "Economics is the science which studies human behaviour as a relationship between ends and scarce means which have alternative uses" (Robbins, 1932:15). For Robbins (1932:21), the subject matter of economics was human behaviour under conditions of scarcity, rather than 'the causes of material welfare' which Robbins interprets to be the meaning of former definitions. Robbins (1932:21) contended that it is not the substance (the materiality) of goods (i.e., use value) that gives them status as economic goods, but rather "it is their relation to valuations…their form…which is significant".

On scarcity, Robbins (1932) explained that it is not an 'infrequence of occurrence' that defines scarcity – an absolute quality – but rather a limitation in relation to the demand. Congruently, "wealth is not wealth because of its substantial qualities. It is wealth because it is scarce" (Robbins, 1932:46). For Robbins, wealth was a relative concept, not an absolute quantity. In this sense, if demand changes, so too do 'scarcity' and 'wealth'. Whether a good or service is an economic good depends on its supply in relation to its demand. Robbins wrote, "There is no quality in things taken out of their relation to men which can make them economic goods. There is no quality in services taken out of relation to the end served which makes them economic" (Robbins, 1932:46). This leads Robbins to the statement that "Value is a relation, not a measurement" (Robbins, 1932:56), value is conceived as "an expression of an order of preference" (Robbins, 1932:59), as such "comparisons of prices have no precise significance, unless exchange is possible between the commodities whose prices are being compared" (Robbins, 1932:59).

Elaborating on the 'ends – means' element of the definition, Robbins (1932:23) argued that the 'ends' are not a concern of economics, but rather how the attainment of ends is limited: "The ends may be noble or they may be base. They may be 'material' or 'immaterial'…but if the attainment of one set of ends involves the sacrifice of others, then it has an economic aspect" (Robbins, 1932:24). Economics takes the 'ends' as given and is concerned only with the allocation of means towards the given 'end'. For Robbins (1932:135-6 and 140), economics is:

> "incapable of deciding as between the desirability of different ends. It is fundamentally distinct from Ethics…Faced with the problem of deciding between this and that, we are not entitled to look to Economics for the ultimate decision. There is nothing in Economics which relieves us of the obligation to choose… it [economics] serves for the inhabitant of the modern world with its endless interconnections and relationships as an extension of his perceptive apparatus. It provides a technique of rational action".

From the 1950s economics became associated with the analysis of decision-making and a particular set of tools, rather than a specific subject matter (Sent, 2006; Backhouse and Medena, 2009). However, not all economists supported the extension of economics to all areas of 'choice'. Commenting on the 'Robbins' definition, Coase (1978:207) contended that "[s]uch a definition makes economics a study of human choice. It is clearly too wide if regarded as a description of what economists do. Economists do not study all human choice". Economists study "the working of the social institutions which bind together the economic system: firms, markets for goods and services, labour markets, capital markets, the banking system, international trade, and so on" and adopt a specific theory of human behaviour – that of humans as rational, utility maximisers – to address social problems (Coase, 1978:207). However, wrote Coase (1978:208):

> "it by no means follows that an approach developed to explain behaviour in the economic system will be equally successful in the other social sciences". In these different fields, the purposes which men seek to achieve will not be the same, the degree of consistency in behaviour need not be the same and, in particular, the institutional framework within which the choices are made are quite different".

Coase (1978:209) commented on the success of economists in moving into other social sciences and attributes (part of) the success to the economic approach of considering the working of the system as a whole; economists are, therefore, "more likely to uncover the basic interrelationships within a social system". The main advantage economics brings to the study of other social systems is "simply a way of looking at the world" (Coase, 1978:209). A similar sentiment is expressed by Pagiola et al. (2004:38) who suggest that just by asking the questions involved in an economic valuation we can illuminate the role of ecosystems with a broader socio-ecological system and thus an important benefit of economic analysis is that it "forces us to grapple with our limited understanding of ecosystem processes and the way they affect human welfare".

Contemporary mainstream economics (neoclassical economics) is underpinned by the Robbins 'scarcity' definition of economics (i.e., the study of the allocation of scarce resources across alternative uses). However, the assumptions of neoclassical economics have come to be extensively questioned from various directions both within the economics discipline and from without. Both the scope of economics, and the boundary or realm of the economic system, are topics of current debate. Increasingly, economic and social systems are recognized as interrelated sub-systems of a global ecological system (Costanza et al., 1997a), while many social problems (wicked problems) are considered indefinable and non-separable (Rittel and Weber, 1973) and cannot be adequately addressed from the perspective of individual disciplines (Max-Neef, 2005; Batie, 2008). Economic analysis will require major modifications to be successfully applied to social problems beyond the 'economic system' contended Coase (1978). The next section presents a brief reflection on the ontological foundations of neoclassical economics as grounds for considering the meaning of 'economic value'.

2.1.2.2 An economic world view

The development of neoclassical economics in the 19th century was strongly influenced by the natural sciences and the associated assumptions of the positivist paradigm. More specifically, the foundations of neoclassical economics are rooted in a mechanistic ontology adopted from the physical sciences (Fullbrook, 2008; Potts, 2010). The mechanistic view of the world developed during the renaissance and is associated with a reductionist perspective - the belief that the 'whole' can be decomposed into parts and that every phenomenon can be analysed (understood) by breaking it down to its fundamental (indivisible) elements, which can be analysed individually (Dopfer, 1988). Causality is a second core principle of a mechanistic ontology and is the view that all phenomena can be described through cause-and-effect relationships (Dopfer, 1988; Liening, 2013). A mechanistic ontology reflects a deterministic world view (Liening, 2013) - the view that each event can be determined by the events that preceded it – and implies that, given complete knowledge of the current state of the 'world', we should be able to predict (determine) the future state. The classical mechanics view presents the living world as consisting of rationally determined human beings (Liening, 2013). Mechanistic thinking and the methods of physics continued to influence economics into the 20^{th} century (Fullbrook, 2008).

Mechanistic influences are apparent in several areas of economic theory. Dopfer (1988:686) suggests that the causality principle of classical physics "represents in general the paradigmatic core of modem economics, particularly in its neoclassical version". Mathematical ideas and methods from mechanics became a significant aspect of neoclassical economics (Liening, 2013). Another example reflecting the absorption of ideas from mechanics into economics is seen in partial analysis and the *ceteris paribus* assumption – which implies a given state can be subdivided into elements (variables) which can then be analysed individually, while 'holding all other things (variables) constant'. Marshall (1890: Book V, Chapter V:2) captured the essence of the *ceteris paribus* assumption:

> "The study of some group of tendencies is isolated by the assumption *other things being equal*: the existence of other tendencies is not denied, but their disturbing effect is neglected for a time. The more the issue is thus narrowed, the more exactly can it be handled: but also, the less closely does it correspond to real life. Each exact and firm handling of a narrow issue, however, helps towards treating broader issues, in which that narrow issue is contained, more exactly than would otherwise have been possible. With each step more things can be let out of the pound; exact discussions can be made less abstract, realistic discussions can be made less inexact than was possible at an earlier stage".

A mechanistic perspective is also apparent in the *homo oeconomicus*[9] model of human behaviour, particularly in the assumptions of rational behaviour and the reduction of all actions to individual preferences.

While the influence of mechanistic thinking on economics is still prevalent, these foundations are increasingly questioned. The adoption of mechanistic principles in economics is

[9] See section 2.1.2.6 for a discussion of the *homo oeconomicus* model of human behaviour.

challenged from two key perspectives: (i) that the physical sciences have since shifted away from a mechanistic worldview (Dopfer, 1988; Leining, 2013), suggesting that the classical mechanical paradigm is no longer a reliable framework; and (ii) that the core principles of 'classical mechanics' are not appropriate for the social sciences. Maki (2001), for example, argued that within the social sciences the interactions between 'causes' are significant which challenges the reductionist assumptions of a mechanistic ontology and questions the reliability of partial analysis.

Ekstedt and Fusari (2010) suggested that economic philosophy is in an era of transformation and development. Based on a historical analysis of the economic discipline, Davis (2006:17) argued that neoclassical economics has, or is in the process of being, displaced by 'a new mainstream pluralism', noting that content from outside fields are increasingly drawn into economic research. Influences from, for example philosophy, ecology, human geography and psychology, have influenced economists and the way people think about the economic system and its relation to other social and ecological systems. Current social issues, such as environmental degradation, population growth, inequality and climate change have challenged the way we make choices, questioned the way we relate to each other and to non-human elements and highlighted the need to reconsider our social goals and value systems. As remarked by Marshall (1920: I, III) "every change in social conditions is likely to require a new development of economic doctrines". De Wit (2019), in questioning whether the 'mind of economics' is shifting, argued that a shift is apparent, but concluded that its effects are "mostly theoretical" and "[t]heir full effect on economic policymaking still has to become manifest" (de Wit, 2019: 251).

While critiques of neoclassical economics and emerging perspectives have not, as yet, led to a single reformulation of the foundations of mainstream economic analysis, theoretical developments have emerged in the form of various schools of thought such as behavioural economics, ecological economics and institutional economics. Spash (1999) argued that, in the development of neoclassical economics, the ethical roots of economics became secondary to the engineering or mechanistic aspect and proposed that ecological economics aims to reintroduce ethics as an integral part of economics. The next section turns to the evolving role of nature in economics.

2.1.2.3 *The role of nature in economics, an evolution*

The role and 'value' of nature in human economies has changed over time. In ancient and medieval economies - pre-classical economics - land was important as a source of agriculture and other extractive industries. In the Mercantilism view (mid-16th and late 17th centuries), the value of land lay in its capacity to feed a growing population and as a source of precious metals (Hubacek and van den Bergh, 2006). The Physiocrats (18th century), a group of French social philosophers, saw agriculture as the primary source of wealth. Economic output was linked to the amount of arable land available, and the value of land was derived from the value of agriculture (Blaug, 1985; Costanza et al., 1997b).

Classical economic thinking emerged towards the end of the 18th century, with a focus on economic growth, economic freedom and deriving the factors of production. Land, labour and (financial) capital were considered the triad of production factors (Hubacek and van den Bergh, 2006) and classical economic thinkers maintained an appreciation of the value of land as a means of production (Gómez-Baggethun et al., 2010). How classical economists perceived the intangible benefits of nature is less clear. Gómez-Baggethun et al. (2010) suggested that some classical economists recognized nature's 'productions' or 'services' in relation to their value in use (in contrast to value in exchange). Spash (1999) argued that some of the Victorian economists of the 19th century, particularly John Stuart Mill, recognized both the potential of non-renewable resources to constrain economic growth and the consequences of unrestrained economic growth for natural ecosystems.

The attention given to land and nature by the classical economists diminished towards the end of the 19th century with the emergence of neoclassical economics. Several shifts in economic thinking took place at this time which influenced neoclassical economic thinking on land and nature including: (i) a shift in focus on land and labour as factors of production to one of labour and capital (Hubacek and van den Bergh, 2006); (ii) a move away from physical measures of capital to monetary and similarly aggregated measures of capital (Hubacek and van den Bergh, 2006); and (iii) a focus on exchange value, and its determinants, with the onset of the marginal revolution in the 1870s (Foster and Clark, 2009). The marginalist era was characterized by a shift towards the tenets of individualism and utility maximisation, as captured by the *homo oeconomicus* model of human behaviour, and an emphasis on mathematical formalism. Concerns in economics about conservation issues during this stage tended to focus on 'wise use' (rather than preservation) and were generally the scope of sub-fields such agricultural economics and not the concern of mainstream economics (Spash, 1999).

Over time, and strongly motivated by the environmental movement of the 1960s, criticisms of neoclassical economics and its ability to address environmental concerns emerged (Røpke, 2004). At the same time, thinking on the environment and sustainability was developing in other disciplines. Several of these developments, particularly those related to systems theory, thermodynamics and the study of ecosystems in terms of energy and material flows, and an increasing interest in inter- and trans-disciplinary research, had a powerful influence on some economists and other researchers involved the environmental field (Costanza et al., 1997b; Spash, 1999). For Costanza (2000:4) this reflected a turning point:

> "The evolution of the human economy has passed from an era in which human-made capital was the limiting factor in economic development to the current era in which remaining natural capital has become the limiting factor".

Barbier and Heal (2006) regarded the framing of the natural environment as natural capital as the basis of an emerging paradigm concerned with the identification, analysis and valuation of the ecosystem services that flow from natural capital. Gomez-Baggethun and de Groot (2010) suggested the concepts of natural capital and ecosystem services are an articulation of a new form of understanding in economics. Spash (1999) argued that 'naming nature as capital' is in line with a mechanistic approach, and reduces natural systems to production systems serving human ends. The following sub-sections turn to various theories and concepts underpinning the neoclassical economics interpretation of value.

2.1.2.4 Value in use, value in exchange

'Value in use' and 'value in exchange' are distinct concepts of value in economics. Use value, an absolute concept, is an expression of the utility of an object related to its physical properties and some element of 'intrinsic value' (Marshall, 1920). The use value of an object is determined by its own properties and the subjective importance of the object to the individual and can be related to both physical and symbolic satisfactions (Spangenberg and Settele, 2016).

Exchange value, in contrast, is relative, reflecting the relation between two objects at a particular time and place in terms of the (exchange) power possession of one object conveys to obtain (purchase) other objects (Marshall, 1920). The exchange value of an object is determined in relation to another object (or set of objects), based on the subjective preferences of the agents involved (Spangenberg and Settele, 2016). The basic assumption of exchange value is that the agents are willing to trade (to receive or to give up) which implies that different sources of value are potentially substitutable (Heal et al., 2005).

A common illustration of these two expressions of value is the diamond-water paradox. Water, being essential for life, has a high, even indefinite use value, yet it has a low value in exchange; whereas diamonds, which are of little practical use, carry a high exchange value (Farber, 2002; Nicholson and Snyder, 2012). The paradox is that some essential things (e.g., water) can have a low exchange value relative to non-essential items (e.g., diamonds). The paradox stems from the relationship between exchange value and scarcity; for something to have value in exchange, it must be perceived as both 'useful' and scarce (Blaug, 1985). Water is perceived as abundant and diamonds as scarce.

In classical economics, nature and its 'services' were viewed as holding use value, but, given their perceived abundance (and non-excludable characteristics), held little value in exchange (Gómez-Baggethun et al., 2010). With the neoclassical focus on exchange value, attention to use value fell away and economic 'value' became synonymous with a theory of exchange value (and thus price) which had resulting implications for the perception of the value of nature and its 'services'. Wealth, while originally associated with use value, came to be defined solely in terms of value generated through exchange (Foster and Clark, 2009); 'nature' then, played little role in defining wealth. A view, argued Foster and Clark (2009), which excludes the "possibility of a broader ecological and social conception of wealth". For Foster and Clark (2009), the distinction between use value and exchange value, and particularly how wealth is defined in respect to these values, underpins what the authors described as "the ecological contradictions of the prevailing economic ideology" (Foster and Clark, 2009:2).

2.1.2.5 Marginal value

Exchange value is measured as marginal utility - the desire for one additional unit (of an object / phenomenon) assuming some stock of the object (Farber et al., 2002; Brouwer et al., 2013). The intensity of desire for an additional unit declines with successive units of the object - the theory of diminishing marginal utility – which reflects the 'scarcity' underpinning of exchange value. The 'scarcer' the object of desire is, the greater its (exchange) value on the margin.

Marginal (exchange) value measures the utility associated with a flow of goods from a stock, but does not reflect the (total) utility provided by the stock (Spangenberg and Settele, 2016). An indication of the value of the total utility of the stock (total value) is estimated by aggregating the (marginal, exchange) values of flows overtime assuming that external conditions, particularly the 'scarcity / abundance' of the stock, hold constant or are relatively unchanged (Spangenberg and Settele, 2016). Calculating the total value of an ecosystem (e.g., an entire wetland compared to one additional hectare of wetland) would mean estimating the value derived from the ecosystem compared to the case without the ecosystem (in contrast to the case of with and without an additional hectare of wetland) (Verdone, 2015); a challenging or even nonsensical task (depending on the scale) as it would require predicting what it would mean to be 'without' the ecosystem (Pendleton and Baldera, 2010).

From a neoclassical economic perspective, 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, this entails establishing the incremental change in outcomes (benefits / services) with the restoration and the value of the change (Verdone, 2015). For ecosystem restoration values to be incorporated or accepted into the neoclassical expression of value (marginal, exchange value), there must be a well-defined change in the ecosystem and a similarly well-defined change in human welfare (utility) resulting from the restoration intervention that a 'consumer' can react to, in either a real or hypothetical market (Bockstael et al. 2000; Pagiola et al. 2004; Norton & Noonan, 2007; Pendleton and Baldera, 2010). The marginal approach requires that the 'good' and the 'next unit' be precisely defined, a practical challenge in the case of ecosystem changes (Norton and Noonan, 2007).

The measurement of exchange value on the ‘margin’ is based on small, non-critical, changes in stock (i.e., changes are neither dramatic nor non-reversible) (Turner et al., 2003; Farley, 2012). This assumption has particular significance in the context of ecosystems, which are characterized by ecological thresholds and regime changes (Crepin et al., 2012; Pelenc and Ballet, 2015). For marginal analysis to hold true, the ‘next unit’ should not be capable of tipping the ecosystem over a functional threshold (Pagiola et al., 2004; Barbier, 2007; DEFRA, 2011) and 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).

Beyond a non-critical range or at an ecological threshold[10], 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). Farber et al. (2002:386) distinguish between ecological and economic thresholds, suggesting that in some cases gradual changes in ecosystem conditions can lead to non-linear changes in economic conditions. An example being the case of a dam or lake being ‘suddenly’ closed after a gradual decline in water quality. Such distinctions depend on whether threshold levels are defined in terms of their potential to provide ‘services’ or benefits to humans or not (Jax, 2014). In some cases, it may be possible to identify threshold points and approximate when they may be reached, for example in the transition from oligotrophic to eutrophic states of water bodies (Heal et al., 2005), in other cases, predicting or even identifying ecological thresholds is more challenging or even impossible (Barbier, 2007, Farley, 2012; Jax, 2014).

Under large-scale changes, marginal analysis is less robust and unlikely to capture the full effect of threshold level changes in the ecosystem (Pagiola et al., 2004; Heal et al., 2005) and may therefore be inappropriate or need to be modified in some way (Barbier, 2007; DEFRA, 2011). Where ecological conditions and threshold levels are uncertain, ecosystem (and

[10] Anderson et al. (2009) defined 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 (see Jax, 2014).

restoration) value estimates become less reliable. Farely (2008) argues that once a critical threshold has been reached, marginal valuation becomes meaningless. A further limitation of marginal analysis is that often, at the margin, ecosystem services / benefits can be substituted by built infrastructure, therefore failing to reflect the 'true' value of the ecosystem stock[11] (Gomez and Barton, 2013).

2.1.2.6 Utility and the rational actor model of human behaviour

Underpinning the (neoclassical) economic view of value is the goal of allocative efficiency, judged in terms of utility (the satisfaction of individual preferences), and the rational actor model of human behaviour, conceptualized (hypothetically) as *Homo oeconomicus*. *Homo oeconomicus* reacts independently, systematically and rationally to maximise their utility in the presence of constraints and changes, considering the consequences and uncertainties associated with all possible actions (Costanza, 2000; Gintis, 2000; Farber et al., 2002; Dietz et al., 2005; Nelson, 2008). In this model of human behaviour, the preferences of individuals are taken as given (known) and stable (Gintis, 2000) and, further, that individuals have a consistent ranking of all the alternatives from which to choose (Hausman & McPherson, 2008). Within the *Homo oeconomicus* framework, human motivation and decision-making are driven by the aim to satisfy individual preferences; other motivators (e.g., freedom, justice, sustainability, rights, community preferences), and ethical notions are considered important, but not the concern of economics (Costanza, 2000; Hausman & McPherson, 2008). Welfare economics, which underpins environmental economics, takes such 'rational behaviour' as producing the best outcomes in terms of efficient resource allocation.

This characterization of the human actor leads to a number of fundamental assumptions, several of which are particularly significant in the context of the value of ecosystem restoration. The rational actor model assumes that there is knowledge about outcomes, that is accessible and can be understood (mentally processed), on which individuals can judge the potential consequences of their choices; individuals aim to (and know how to) maximise their preferences based on this information. However, insights from behavioural economics and psychology, among others, have demonstrated that there are limits to individual information

[11] This discussion is taken up further in Section 2.1.2.7.

processing in terms of information availability, individual cognitive capabilities and time[12] (Kahneman, 2003; Samson, 2016). This is certainly the case in the context of ecological restoration. Ecosystems are inherently complex systems, dynamic, unpredictable (Gunderson and Light, 2006; Holling, 1996) and subject to regime changes (Crepin et al., 2012). Accordingly, ecosystem restoration outcomes are variable, dynamic and characterized by uncertainties and limited information (Suding, 2011). The very nature of ecosystems makes the task of comprehending and predicting the outcomes and consequences of restoration challenging even for ecologists (Daily et al., 1997; Bockstael et al., 2000) and few outside the field are likely to be fully aware of the consequences of ecosystem change for humans (Naess, 2006a). By definition (i.e., the assumption of unbounded rationality), meaningful economic values cannot be attributed to, or elicited for, ecosystem change if humanity does not (yet) fully comprehend the resulting outcomes and utility impacts (Bockstael et al., 2000). Bockstael et al. (2000:1388) argued, however, that "this does not mean we should abandon the effort of measuring well defined economic values for those consequences we do understand".

Studies from behavioural economics, experimental economics and game theory demonstrate further, that there are other human behaviour motivators such as fairness, reciprocity, moral duty, and community or cultural norms (Howarth, 1995; Henrich et al., 2001; Gowdy, 2004). Prospect theory (Kahneman & Tversky, 1979), a model of choice under risk, posits that decisions (and valuations) are influenced by context including the way choices are presented, and are made relative to an individual's point of reference (e.g., current wealth) (Kahneman, 2003). These insights suggest that individuals make irrational choices, and challenge the central assumptions of utility maximisation, stable preferences, rational expectations and optimal information processing (Jolls et al., 1998). On the other hand, Nelson (2008:458) suggests that few practising economists consider typical human behaviour to fall solely within the *Homo oeconomicus* framing, but rather that "this model of human behaviour is perceived as being the most useful and most rigorously objective starting point for economic analysis".

[12] In behavioural economics this concept in known as Bounded Rationality, proposed by Herbert Simon (1982).

Within this model, the aggregation of individual preferences is taken as a reflection of social value (Naess, 2006a). Findings in behavioural psychology, neuroscience and social anthropology challenge this assumption by showing that individual choice is influenced by, and embedded in, a social context (Parks and Gowdy, 2013). Individuals do not take decisions in isolation; they comprehend and act from within a culture, that is, from a place of shared values (e.g., corporate values, family values, cultural values) which influence, and can be different from, their personal values (Klamer, 2003). In this way, individuals can hold social preferences separate from personal preferences; an individual may 'act' as both a citizen and a consumer (individual) and these actions may differ (Turner et al., 2008).

Based on the *Homo oeconomicus* model, the basic notion of value in neoclassical economics is anthropocentric and utilitarian (de Groot et al., 2002; Pascual et al., 2010; Spangenberg and Settele, 2016). There is an extensive literature arguing that the neoclassical economic model is limited as an explanation of human behaviour and too restrictive in its conceptualisation of value (e.g., Sen, 1977; Klamer, 2003; Stuhr, 2003; Naess, 2006a; and cultural economics, feminist economics, behavioural economics) including as a way of articulating environmental values (Martinez-Alier et al., 1998; Spash, 1993; Costanza, 2000; Norton and Noonan, 2007; O'Neill et al. 2008; Liu et al., 2010; Chan et al., 2012a, b; Luck et al. 2012; Jax et al. 2013; Kallis et al., 2013; Parks and Gowdy, 2013; Gómez-Baggethun et al. 2014; Doak et al., 2014; Fanny et al., 2015; Spangenbery and Settele, 2016; Arias-Arevalo et al., 2018). On the one extreme is the complete rejection of any economic value of nature, Mark Sagoff for example (Sagoff, 2008); for others (both ecologists and economists), attributing economic value to ecosystems is viewed "as a pragmatic and transitory short-term tool to communicate the value of biodiversity using a language that reflects dominant political and economic views" (Gomez-Baggethun and Ruiz-Perez, 2011:614), and as a way of informing decision-making (Turner et al., 2003; de Groot et al., 2006; Bateman et al., 2011).

Some argue that the economic view of value is conceptually much broader than is often recognized and can capture human-preferences for a range of benefits and value types, including those that have no market value (MA, 2003; Heal et al., 2005; Turner et al., 2008), as indicated in the total economic framework (section 2.1.2.9), and, to a degree, social or group values (Hansjürgens et al., 2017). Others contend that the scope of economic measurement is limited and is only appropriate in certain situations or under specific conditions (Bockstael et al., 2000; Spangenberg and Settele, 2010, 2016; Kallis et al. 2013; Fanny et al., 2015). Bockstael et al. (2000) argued that 'economic value' is not intended to capture all values and is not the only, nor always the most relevant, concept of value (e.g., Bockstael et al., 2000), but that "when the stated objective is to measure economic value, the concepts and principles associated with performing the task are not matters of opinion" and the results must be correctly interpreted within the 'meaning' of economic value (Bockstael et al., 2000: 1386). For Norton and Noonan (2007:665) the economic model is "one of the many metaphors necessary to comprehend the complexities of environmental changes and their impacts on humans" and does not capture the range of values associated with the natural environment. The idea then is "not to stop thinking economically" but to "start thinking in addition to economic analysis" (Norton and Noonan, 2007:672).

For many ecological economists, a fundamental goal is to widen the conceptualisation of human behaviour and economic value beyond the neoclassical economic perspective (e.g., Costanza, 1989; Costanza and Folke, 1997; Spash, 1999; Costanza, 2000). For example, Spash (1999) described ecological economics as an opportunity to reintroduce ethical elements and moral values into economics, in contrast to the 'engineering equation' focus of neoclassical economics. Costanza and Folke (1997) and Costanza (1991, 2000) proposed an expanded 'ecological economic' model to include social goals of 'sustainability' and 'fairness' in addition to 'economic efficiency', adding to the H*omo oeconomicus* model of human behaviour H*omo communicus,* based on the goal of fairness and community preferences, and H*omo naturalis,* based on the sustainability goal and whole system preferences (Costanza, 2000). Since their writing in the late 20th century, sustainability and equity goals have received considerable attention expressed initially in the United Nations Millennium Development Goals and subsequently the Sustainable Development Goals to which multiple countries have expressed commitment to achieving.

2.1.2.7 Weak and strong sustainability

There are multiple approaches to sustainability (Dietz and Neumayer, 2007; Davies, 2013). For economists, 'capital' is a core concept and the sustainability discourse is often framed in terms of 'capital' and related accounting terms such as stocks, flows and return on investment. In essence, ‘capital’ is a stock from which goods / services / benefits flow and represents the productive capacity – in terms of utility satisfaction – of an entity (Klamer, 2002; Ekins et al., 2003). Welfare, or utility satisfaction, is conceptually embodied in several forms of capital – manufactured (physical, produced), human, social (institutional), natural and cultural capital (Throsby, 1999; Ekins et al., 2003) – and the sustainability goal is to maintain capital (stocks) in such a way as to ensure non declining welfare in perpetuity (Gutes, 1996; Dietz and Neumayer, 2007). The core debate underpinning the ‘capitals’ approach to sustainability is whether, and to what extent, different forms of capital are substitutable. At the two ends of the continuum are the weak and strong sustainability positions.

The weak sustainability approach assumes that different forms of capital are essentially substitutable in providing welfare, particularly manufactured and natural capital (Gutes, 1996; Dietz and Neumayer, 2007). The goal, from this perspective, is to maintain, or add to, the aggregate capital stock (Costanza and Daly, 1992; Neumayer, 2012). In this way, it is regarded as possible to compensate the degradation (decline) of natural capital by equivalent amounts of other forms of capital (e.g., manufactured or financial capital) and it is generally assumed that technological progress will generate solutions to the environmental impacts of increased production (Pelenc and Ballet, 2015).

The strong sustainability approach views the relationship between forms of capital, specifically manufactured and natural capital, as one of complementarity rather than (aggregate) substitutability (Constanza and Daly, 1992; Farley, 2012). The goal is to maintain natural capital and manufactured capital separately; that is to maintain total natural capital regardless of changes in the levels of other capitals (Costanza and Daly, 1992). Growing the wealth of one form of capital (e.g., financial capital) must not reduce natural capital.

The strong sustainability view holds that there is a qualitative difference between manufactured and natural capital, and specifically that there are elements of natural capital that are ‘critical’ and their destruction cannot be compensated through investing in the other ‘capitals’ (Ekins et al., 2003; Neumayer, 2012)[13]. Ekins et al. (2003) suggested that in some

[13] Brand (2009) provide a detailed analysis of what constitutes ‘critical natural capital’.

cases substitutability between forms of capital exists; that is, the welfare derived from different 'capitals' are commensurable. This is not, however, always the case and so the authors suggested the prudent approach is to start from a strong sustainability view from which it is possible to shift to a weak sustainability position should it be appropriate. Further, humanity is not, as yet (and may never be), certain about which elements of natural capital are critical and which can be 'lost' or substituted into perpetuity without reducing human welfare (Farley, 2008). By definition, (neoclassical) exchange value implies some degree of substitutability[14] and therefore a weak sustainability position. From a strong sustainability perspective, economic value cannot be considered decisive (Spangenberg and Settele, 2016).

Inherent in a sustainability perspective is consideration of the future. The decisions and actions of present generations have consequences that extend across time. Consideration of temporal scales is especially relevant in the context of ecological restorations that are long term projects, as ecosystems recover over many years (Streever, 1997; Moreno-Mateos et al., 2012), generally with large costs in the short term and benefits extending into the distant future. Discounting is the economics approach to comparing values arising at different points in time and is discussed in the next section.

2.1.2.8 Time, values and discounting

In neoclassical economics, the comparison of values that arise at different points in time is done through 'discounting' projected future benefits (and costs) to a 'present value'; that is, weighting future consequences relatively less to more immediate consequences (Frederick, 2006). Both the notion of 'discounting' and the rate at which future benefits - especially those that flow from natural capital - should be discounted, are contentious topics within environmental valuation (Fenichel et al., 2017). Intergenerational interdependencies and obligations raise difficult moral questions and judgments on how to evaluate long-term public investments, such as ecological restoration (Spash, 1993; Wesley & Peterson, 1993); economics itself provides no definitive answers.

[14] See Section 2.1.2.4 for a discussion on exchange value.

There are two components to the ‘discounting’ debate: (i) whether discounting is appropriate, and (ii) if discounting is accepted, what rate(s) should be applied. Economists tend to focus on the latter, accepting discounting as a method for evaluating public projects and debating the appropriate discount rate. ‘Discounting’ is often justified by the argument (assumption) that future generations will be wealthier than current generations (the wealth effect) and, further, that technological advancements of the future will ‘allow’ future generations to address environmental harms (Wesley & Peterson, 1993; Gollier, 2010). Discounting is also justified by the individual time preference assumption, that is, that people prefer current utility over future utility (Frederick, 2006; Gowdy et al., 2010), notions of ‘agent relative ethics’ (Beckerman and Hepburn, 2007) and arguments related to capital investment and opportunity cost (Zhuang et al., 2007). Other arguments, particularly in the case of distant future consequences, stem from the uncertainty of considering the indeterminate preferences of (infinite) generations who do not yet exist and therefore the preferences of future generations, which are unknown, should not ‘count’ as much as those of the present generation (Wesley & Peterson, 1993; Frederick, 2006).

Philosophers focus on the moral issue of the present generation's obligations to future generations, drawing arguments from, for example, views on the rights of future humans to essential elements (e.g., oxygen), notions of ‘virtue’ and ‘morals’ (e.g., constraining self-interest or greed out of a sense of community, a desire to be well regarded) and from a recognition of various beliefs on the continuity between past and present and between human beings and non-human beings (Wesley & Peterson, 1993; Diaz at el., 2015). In the context of decisions affecting the environment and natural capital, many of these arguments are justified by the uncertainty associated with ecological thresholds and regime changes and the risk of irreversible effects (Wesley and Peterson, 1993). Gollier (2010) highlighted further limitations of the economic discounting approach in the context of uncertainty, raising the point that discounting is based on ‘knowing’ the economic value of future benefits, which in the case of natural capital and intergeneration timeframes is problematic given that economic value is based on relative scarcity which is unknown in the long run. In addition, values (based on individual preferences) change over time with, for example, changes in knowledge, principles and social and environmental conditions (Diaz et al., 2014).

The multiple perspectives and ongoing debate serve to emphasize that discounting, and the selection of a discount rate, are ethical decisions based on a judgement on intergenerational equity and the obligation of the current generation to the well-being of future humans and non-human entities. In the neoclassical economic application, 'discounting' is based on assumptions about (i) the preferences and behaviour of individuals, typically those of 'Westernized' individuals, (ii) marginal changes and (iii) substitutability between forms of capital, and disregard considerable uncertainty concerning the outcomes of ecosystem change (Wesley and Peterson, 1993; Gowdy et al. 2010). While reliant on significant assumptions, discounting is often applied, and a discount rate selected, with little explicit justification (Spash, 1993; Frederick, 2006). Given that small changes in the discounting procedure and rate can have significant influence on evaluation outcomes, Farley (2012) maintains that the choice of procedure and rate should be explained, and the underlying assumptions made explicit in any economic analysis, while Fenichel et al. (2017) contend that questions about discounting should continue to be debated in the policy arena (Fenichel et al., 2017).

2.1.2.9 Economic value typology

In considering environmental values, economists have identified a broad range of value types commonly organized into the Total Economic Value (TEV) typology (Barbier, 1993; Pearce and Moran, 1994; Dixon and Pagiola, 1998; Turner et al., 2008; DEFRA, 2011). The TEV typology aims to articulate a range of utilitarian values associated with ecosystems, thereby identifying a range of ways human welfare is influenced by changes in ecosystem condition and function. Values within the TEV typology are typically disaggregated into 'use' and 'non-use' values; further sub-divisions of these categories and terminology tend to vary between analysts.

Use values arise from the actual use of a given resource and provide direct sources of utility (Pearce and Moran, 1994; Hanley and Barbier, 2009). These may be 'direct use' values, which derive from the interaction of an agent with the ecosystem or its outputs; or 'indirect use' values, which arise from the regulatory functions of ecosystems that affect individuals, and do not depend on direct interaction with the ecosystem (Barbier, 1993; Turner et al., 2008). In the case of wetlands, reed harvesting (consumptive) and bird watching (non-consumptive), for example, provide direct use values. The assimilation of pollutants from water or the stream flow regulation services provided by wetlands are associated with indirect use values. ‘Option value’, often included as a category of use values, arises from the ‘option’ to use, or benefit from, the ecosystem in future even if these benefits are not currently realized (Turner et al., 2008). Option values stem from uncertainty regarding the demand for a certain use (benefit) and / or its availability (supply) in the future (Barbier et al., 1997) and reflect an agent's desire to secure an asset for the option of using it in the future (Pearce and Moran, 1994). Option value is ‘likened’ to insurance value (Dixon and Pagiola, 1998). The inclusion of ‘option value’ within the TEV typology has been contested (Hanley and Barbier, 2009). An alternative perspective is that ‘option value’ is a way of framing TEV under conditions of uncertainty, or as the value of delaying a decision/action where there is potential for ‘new’ knowledge to emerge (known as quasi-option value) (Pascual et al., 2010).

Non-use value is derived from the knowledge that the natural environment is preserved independent of any use (Crowards, 1997; de Groot et al., 2006; DEFRA, 2011; Baum, 2012). In contrast to use values, non-use values are far less tangible meaning they are more difficult to define and categorize, and subject to debate. Typically, non-use values are disaggregated into the categories of existence, bequest and altruistic/philanthropic value (Crowards, 1997; Turner et al., 2008). Existence value refers to the value people place on the continued existence of a species, ecosystem or feature (Krutilla, 1967). Existence value may be motivated by a concern for the object (e.g., a threatened wetland bird) or out of a sense of responsibility or stewardship (Turner et al., 2008). Bequest value captures the value attached to preserving ecosystems and biodiversity for the enjoyment of future generations (Mitchell and Carson, 1989; Barbier et al., 1997), while altruistic value is associated with the availability of ecosystems and biodiversity for the enjoyment of contemporaries in the current generation (Aldred, 1994).

While conceptually the TEV typology encompasses a broad range of values, Turner et al. (2008:42) suggested that some ‘values’ included as non-use values are “arguably outside the scope of conventional economic thought”. Similarly, Crowards (1997) questioned whether the underlying motivations of (some) non-use values make them incompatible with economic evaluation. Crowards (1997) argued, as an example, that altruistic value can be evaluated within an economic framing in the case where the ultimate motivation is (or is assumed to be) self-interest (i.e., individual preference satisfaction); whereas truly selfless altruistic motives are not compatible with economic analysis[15]. For Hansjürgens et al. (2017:11), the TEV typology is “a heuristic to consider different value dimensions and to acknowledge the fact that individuals hold values for a good or service because of very different motives” and should not be viewed as an accounting framework for aggregating different values into a ‘total’ value. Turner et al. (2008:46), specifically with reference to wetlands, emphasized that the TEV framing is not “an exhaustive assessment of the value of wetlands to society…[t]here are other sets of values that are supplementary to total economic value”. Turner et al. (2008) gave as examples, ‘contributory value’ which relates to the contribution of wetland ecosystems to biodiversity, and values associated with the role of wetlands in a broader socio-ecological system in terms of stabilizing natural systems and protecting and supporting built infrastructure and economic systems.

In practice, the ‘categories’ of TEV and their underlying motivations are likely to overlap and be difficult to disentangle and measure separately (Mitchell and Carson, 1989) and the application of the TEV typology is often restricted to those aspects that it is possible and feasible to quantify, resulting in ‘partial’ analysis and further limiting TEV as a reflection of the value of an ecosystem (Turner et al., 2008).

2.1.3 Concepts from the ecosystem valuation field

In the field of ecosystem valuation, value types are regularly classified into three broad groups: economic, cultural and ecological values. With regard to ecological restoration, Clewell and Aronson (2007) argued that four sets of values, namely ecological, personal, socio-economic and cultural, are fundamental within a holistic model of restoration, emphasizing that restoration plays a positive role in multiple areas of human life. More recently, additional environmental value types or categories have been recognised including spiritual, symbolic, place, shared and relational values (e.g., Díaz et al., 2015; Kenter et al., 2015; Chan et al., 2016; Arias-Arévalo et al. 2018; Chan et al., 2018). Several of these value types are introduced in the following sub-sections.

[15] This argument relates to the deeper question of the existence of ‘truly selfless motivations’ and links to the ongoing debate within environmental philosophy and ethics on non-anthropocentric values (section 2.1.1.3).

2.1.3.1 Shared, social and cultural values

The concepts of 'shared' 'social' and 'group' values increasingly appear in the environmental valuation literature albeit to represent a variety of interpretations, but often in contrast to 'individual' values and /or in association with 'cultural' values. Given that economic value is based on individual preferences and (economic) value to society is taken as the aggregation of individual preferences (not without controversy), it is worth briefly 'unpacking' these 'collective' value perspectives.

The term 'shared' is often used with a normative sense of 'values' as guiding principles and reflects the ideals shared (held in common) by a collective (Kenter et al., 2015). Shared values are not necessarily shared equally by all members (Chan et al., 2012b; Díaz et al., 2015). In the context of ecological restoration, Clewell and Aronson (2007) reasoned that ecosystem restoration is motivated by a goal of satisfying values shared collectively within a culture, in contrast to the satisfaction of individual preferences. While individuals can hold values at both a personal and a collective level that differ (Kenter et al., 2015); individual and shared values are not mutually exclusive, but rather mutually constitutive (Hodgson, 2003). The values individuals hold originate from, or are at least influenced by, the social context or culture(s) in which they live and have lived (Klamer, 2003; Parks and Gowdy, 2013). At the same time, shared values are shaped and influenced by the individuals of the collective.

'Social value' can refer to the shared values of a society (as above). It is also used in contrasting 'value to society' to 'value to an individual' where value to society is "the benefit, worth or importance of something to society as a whole" (Kenter et al., 2015:88). In this sense, 'social value' is linked to social scale, public interest, and the value of public goods and social processes (Kenter et al., 2015). Kenter et al. (2015) distinguished 'shared value' as describing the nature of those holding value (i.e., the collective) and 'social value' as referring to a type of value.

'Cultural value' is similarly a variously used and often disputed term. Chan et al. (2012a:14) captured this sense by describing the various 'classes' of values grouped as cultural values within the environmental values field as "best understood as those that do not fit well in other sectors". 'Cultural value' can refer to the values shared by a culture (e.g., Throsby, 1999), in this sense, values are associated with the ideals of a collective. An object or phenomena has cultural value if it contributes to the fulfilment of these shared ideals (Throsby, 1999).

In the environmental valuation field, 'cultural value' is more commonly (but not exclusively) associated with the worth, or relative importance, of goods/services/benefits 'classed' as cultural. Chan et al. (2012a:9) defined cultural services broadly as "ecosystems' contributions to the non-material benefits (e.g., capabilities and experiences) that arise from human–ecosystem relationships". In their discussion, the authors argued that the terms 'values', 'benefits', and 'services', while often conflated, are distinct: 'services' are the ecosystem processes underpinning 'benefits', which are goods and experiences of value to people. Similar arguments are put forward by other theorists for example Boyd and Banzhaf (2007), Fisher et al. (2009) and Mace et al. (2011). Benefits are more easily recognized by people than the services from which they arise (Chan et al., 2012a) and, further, numerous benefits can stem from a single service; and a single benefit can hold multiple types of value (e.g., cultural and economic value).

2.1.3.2 Ecological value

Ecological value is a much debated concept, particularly with regard to being defined in terms of instrumental or intrinsic value. Following Farber (2002) and de Groot et al. (2002), ecological value reflects the degree to which an ecological component or process contributes to ecological sustainability - maintaining life on earth. The degree of ecological value is expressed through attributes or indicators such as ecosystem integrity (health) and resilience. The ecological value of restoration then, is the contribution of the restoration to the recovery of a degraded system to an intact condition in terms of its ecological attributes of integrity and resilience (Clewell and Aronson, 2007). In the context of environmental valuation, de Groot et al. (2010:23) cautioned that ecosystem integrity and resilience, as measures of value (importance) should be "distinguished from what can be included in economic values because although they contribute to welfare, they cannot readily be taken into account in the expression of individual preferences, as they are too indirect and complex, albeit they may be critical for human survival".

Turner et al. (2008:46), in referring to wetland ecosystems, identified several sets of 'ecological' values described as "supplementary to total economic value". For example, 'contributory value' relates to the contribution of wetland ecosystems to biodiversity, and values associated with the role of wetlands in a broader socio-ecological system in terms of stabilizing natural systems and protecting and supporting built infrastructure and economic systems (Turner et al., 2008). Contributory value emphasizes the complex relationships of interaction and interdependence between elements and species within ecosystems and, therefore, the limited substitutability within the web of biodiversity. Turner et al. (2008) highlighted the importance of the structure and functioning of systems from which 'services' flow and introduced the concept of primary and secondary value as illustrative. As explained by Turner et al. (2008:47):

> "The primary value describes the system characteristics: the self-organizing capacity of the system including its dynamic evolutionary processes and capacity to absorb external disturbances. It relates to the aspects of the system that 'hold everything together' and is consequently also referred to as 'glue value'. Secondary value refers to the renewable flow of benefits generated by the natural system. It is dependent on the continued operation, maintenance and 'health' of the system as a whole".

2.1.3.3 Relational value

The concept of 'relational value' has emerged in the field of environmental valuation to bridge a perceived gap in the usual framings of 'instrumental' and 'intrinsic' value to reflect the multitude of human-nature relationships (Díaz et al., 2015; Chan et al., 2016; Arias-Arévalo et al., 2018). Relational value is posited as a means of expressing values that emerge from the relationships people hold with nature beyond both the tangible 'benefits' they derive from ecosystems and the inherent worth of nature (intrinsic value). These relationships include relationships between people that involve nature such as the impact of water pollution (Chan et al., 2016).

'Relational value', as applied to environmental values, is an evolving concept (Stenseke, 2018), but has initially been conceived as the preferences, principles and virtues that derive from human relationships with, and responsibilities towards, nature (Chan et al., 2016; Jax et al., 2018; Muradian and Pascual, 2018). Relational models of, for example, 'stewardship' ‘utilization’ and ‘ritualized exchange’ are recognized, along with values connected to the fundamental conditions to support life on earth, contribute to a sense of meaning and identify and promote ‘good human life’ or flourishing (eudaimonistic, including adhering to one’s moral principles and collective flourishing) (Chan et al., 2016; Muradian and Pascual, 2018).

In this view, nature becomes the space where interactions among the ecological, social and cultural take place in a relational way (Muraca, 2011; Chan et al., 2016). This perspective is apparent in the concept of ‘landscape’[16] which is understood as a platform for integrating human society and the natural sphere (Stenseke, 2018). A ‘landscape’ is constituted through its relations to other things; as the relations change the ‘landscape’ changes (Bassett and Peimer, 2015). In this way, landscapes are dynamic, spatially and historically contingent, and subjective, viewed differently by different people (Bassett and Peimer, 2015; Stenseke, 2018).

From a relational perspective, values originate in the spatially and historically influenced relationships and meanings between people and their environments and ecosystems. These values are as numerous as there are human-nature relationships, context specific, and dynamic, evolving as human-nature relationships shift. The diversity of human-nature interactions leads to a continuous shift in the meaning (value) humans attribute to the non-human environment. Adopting a relational perspective requires a focus on the interdependent and dynamic ways in which the biophysical and social interact (Stone-Jovicich, 2015).

2.1.3.4 Value incommensurability

Can the various values of restoring wetlands be compared? Can they be reduced to a common metric (value commensurability), and further, a monetary metric (monetary commensurability)? The comparability of values is a key concept in environmental valuation. Value *incommensurability*, in the context of environmental values, implies a plurality of values that cannot be compared along some common *cardinal* scale such as money or energy (Martinez-Alier et al., 1998; Aldred, 2006). *Incomparability* suggests that values cannot be compared even on an ordinal scale (ranking) (Aldred, 2006). As such, values may be

[16] From social-ecological relations theory, see Bassett and Peimer, 2015.

incommensurable, but still comparable (i.e., ordered relative to one another). Monetary CBA can only capture values that are commensurate to a monetary metric.

A related concept, often raised in the environmental valuation field, is the commodification of nature, that is, reducing environmental values to exchange values (Vatn, 2009) and the introduction of market exchange into previously non-marketed areas of the environment (Gómez-Baggethun and Ruiz-Perez, 2011). Gómez-Baggethun (2010) demonstrated the process of ecosystem function commodification, starting with a utilitarian framing, to monetization, to appropriation and culminating in the exchange of ecosystem services in the market, and suggested that this process had been completed for a number of ecosystem services as demonstrated in the establishment of Payment for Ecosystem Services (PES) schemes.

Value incommensurability and value pluralism are increasingly accepted in social-ecological systems and ecosystem services research and the field of ecological economics (Martinez-Alier et al., 1998; Luck et al., 2012; Gómez-Baggethun and Barton, 2013; Raymond et al., 2013; Díaz et al., 2015; Chan et al. 2016; Kenter 2016; Arias-Arévalo et al. 2018). For example, Gómez-Baggethun and Barton (2013:238) advocate a value pluralism perspective that acknowledges 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". However, several advocates of value pluralism rely on the ecosystem services framing which is utilitarian and anthropocentric (de Groot et al., 2002; Heal et al., 2005) and emphasizes the instrumental value of the environment (Reyers et al., 2012; Muradian and Pascual, 2018) and thus could itself be viewed as a monistic value theory. While value pluralism is increasingly recognized as a principle of ecosystem valuation, Arias-Arévalo et al. (2018) suggest that the operationalisation of value pluralism remains uncommon.

2.1.4 Section conclusion

Drawing from the disciplines of philosophy and economics, and the environmental valuation field, this section presented various concepts and perspectives associated with environmental values. Rather than providing a comprehensive account of value theory, the section emphasises that “value is not a single, simple concept” (EPA, 2009:13). Several points that stand out from the discussion are summarized in sub-section 2.1.4.1. Their implication for the economic valuation of wetland restoration is contemplated in the closing sub-section.

2.1.4.1 A multiplicity of values

The perspectives presented in this section emphasize that value perspectives and values associated with the environment are numerous. There are multiple interpretations or meanings of the term ‘values’ (e.g., principle, preference, measure), different sources of value (e.g., intrinsic, instrumental, relational), and diverse value types (e.g., economic, cultural, ecological). Environmental values can be incommensurable and differ between personal (individual) and group values. Values are not fixed, but emerge within a particular social, political and cultural context, at a specific time, and are influenced and mediated by people’s worldviews and the knowledge and understanding they hold at the time. Values are dynamic; as these ‘contexts’ change, so too can the values people hold or express for the environment shift. Values can change with reflection and deliberation and are influenced by temporal, spatial and social organizational scales.

Economic value is based on a specific philosophy of value; economic values of the environment are instrumental and anthropocentric, embedded in the preferences of individuals. Economic values are also context and scale dependent (spatial, temporal). Importantly, economic value is a relative measure related to exchange and scarcity, and measured at the margin. Inherent in the concept of ‘exchange’ is possession; for a benefit or outcome of wetland restoration to hold ‘economic value’ it must, at least theoretically, be tradable. Many of the attributes of wetlands are public goods, suggesting that economic value may not be the most appropriate measure of value for such attributes. Further, economic value reflects relative scarcity; the economic value of attributes / objects not perceived as scarce will reflect a ‘low’ economic value. The economic value of wetland restoration reflects the contribution to human welfare of the additional ‘flow’ (increased supply) of goods or services as a result of the restoration based on the perceived ‘use’ and scarcity (demand) for the additional units of the goods or services. The economic value of wetland restoration reflects the value of independent elements associated with the structure and function of a wetland.

The values of wetlands, and therefore wetland restoration, are multiple, diverse and interrelated. This multiplicity is not captured within the sphere of economic value alone, which reflects only one way in which humans interact with the non-human world and one 'type' of value. Economic value does not represent the 'value' of restoring a wetland, rather it reflects the magnitude of the restoration's contribution to increasing or enhancing the flow of goods or services that contribute to satisfying individual preferences and are perceived as relatively scarce and tradable. While the economic value perspective can be a useful framework for considering the value of wetland restoration, it is a partial one. In the words of Norton and Noonan (2007:665), economic value is one "of the many metaphors necessary to comprehend the complexities of environmental changes and their impacts on humans".

2.1.4.2 Situating economic value within a multiplicity of environmental values

The fact that economic value can reflect only a part of the value of the environment does not invalidate the economic valuation of wetland restoration. It does mean that economic value should be viewed as one source of information on environmental values, and economic valuation considered as one of the tools available for evaluating a particular action. The diversity of environmental values, does however, compel any 'valuer' (or valuation) to situate the articulated value(s) within a broader conception of values. To express or articulate a particular value entails revealing its place among others.

The classification of environmental values by Arias-Arévalo et al. (2018) offers a way to situate economic value within a broader conceptualization of values. Arias-Arévalo et al. (2018) use three metaphors of the human-nature relationship to classify environmental values and link them to several value domains (instrumental, intrinsic, fundamental and eudemonistic), Table 2.1. While instrumental, intrinsic and relational values are distinguished; the boundaries between domains are not distinct; the domains overlap, are intertwined and co-exist. The classification system is intended as a heuristic tool to support evaluation and emphasize the multiplicity of environmental values.

Taking a nested systems approach, Figure 2.1 illustrates the 'placement' of economic value within the classification and highlights the many values that may be associated with the environment and, therefore, with wetland restoration, and how economic value only captures a part of this network of values.

Table 2.1: Classification of environmental values across different metaphors of human–nature relationships

Human-nature metaphor	Domain	Articulated/assigned value
Gaining from nature Emphasis on tangible elements of nature, economic productivity	*Instrumental* Ecosystems and biodiversity as a means to achieve utility	Economic value
Living for nature The environment is shared by human and non-human species Non-human species deserve concern for their own sake	*Intrinsic* Biodiversity and ecosystems have value in themselves	Moral duties towards nature
Living in nature Collective well-being Sense of 'what is right' Encompasses instrumental value in the sense of human goals, but emphasises sustainability and intangible elements	*Fundamental* Conditions to: i) protect the life-supporting system, ii) allow people to define themselves, and iii) give sense to their existence	Sense of basic needs/survival Ecological integrity & resilience Subsistence Mental and physical health Identity
	Eudaimonistic Entities and processes which represent conditions for leading a 'good human life'	Well-being, good life, flourishing Meaningful occupation Recreation, leisure Cognitive development

Source: Adapted from Arias-Arévalo et al. (2018:38).

The multiplicity of environmental values is increasingly recognized in social-ecological systems and ecosystem services research and the field of ecological economics. In recent years, ecosystem valuation has extended beyond a focus on instrumental value with the integration of intrinsic (or sometimes biophysical or ecological) value, cultural value, shared value and relational value (Díaz et al., 2015; Kenter et al., 2015; Arias-Arévalo et al. 2018; Chan et al., 2018). Integrated valuation, within the environmental valuation field, attempts to operationalise the concept of the multiplicity of values; integrated valuation studies aim to express multiple dimensions and types of values so that decisions can be informed by a diversity of value systems (Gómez-Baggethun et al., 2014). However, while conceptually accepted, the operationalisation of value diversity within environmental valuation is tentative (Arias-Arévalo et al., 2018) and characterised by numerous interpretations of values and value pluralism.

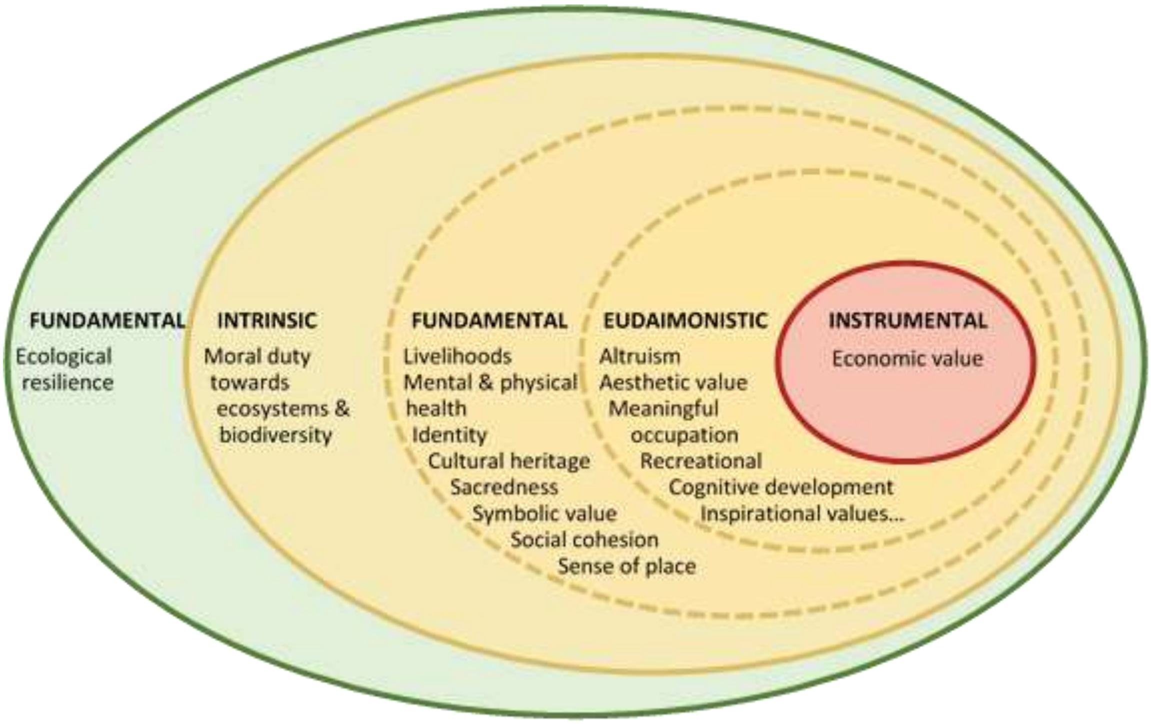


Figure 2.1 Schematic illustrating the 'placement' of economic value within the classification of value domains proposed by Arias-Arévalo et al. (2018).

Source: Reproduced from Arias-Arévalo et al. (2018:41).

The challenge remains to integrate multiple values into a wider framework of environmental valuation. Perhaps a starting point is to raise awareness and create space for the consideration of other values within the economic valuation process, which is far more developed in practice, and to situate articulated economic values within the multiplicity of environmental values. Fundamental to achieving such an objective is to consider (and articulate) the context within which the valuation is made.

2.2 WETLANDS, WELL-BEING AND RESTORATION

"When we try to pick out anything by itself, we find it hitched to everything else in the Universe" (Muir, 1911:110).

Wetland restoration is a globally accepted response to the degradation of wetlands that aims to steadily increase the extent and integrity of wetland ecosystems. Drawing from the literature, this section introduces various aspects related to the restoration of wetland ecosystems, including the definition of wetlands and wetland restoration and the role of wetlands in sustaining human well-being. Definitions and approaches adopted in the South African context are highlighted. The ecosystem services framing is discussed as an approach for articulating the relationship between the structure and functions of wetlands and human well-being. In concluding the section, several points particularly relevant to the economic valuation of wetland restoration are summarized.

2.2.1 Wetland ecosystems

2.2.1.1 Definition

A wetland is a distinct inland ecosystem driven by the presence of water, either where the water table is near the surface of the land, or where the land is covered by water (Ramsar, 2016). A wetland is a region of transition between a terrestrial and an aquatic system characterized by specific hydrological conditions which lead to unique soil characteristics and biota specifically adapted to saturated soils and wet conditions (Mitsch and Gosselink, 2015). Wetland ecosystems comprise both the abiotic aspects of the area (e.g., water supply, climate and geology) and the biotic community of the wetland (Macfarlane et al., 2008).

Formal definitions of wetlands have been developed, particularly for legal purposes, however wetlands are recognized as diverse and variable across a broad range of hydrologic conditions, locations, sizes and level of human influence (Mitsch and Gosselink, 2015). The intergovernmental treaty 'The Convention on Wetlands' (Ramsar, Iran, 1971), defines wetlands as:

> “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres” (Ramsar, 1971: Article 1.1).

The Convention recognizes 42 types of wetlands belonging to three broad categories: Marine and Coastal Wetlands, Inland Wetlands, and Human-made Wetlands. This is a broad definition of wetland ecosystems essentially encompassing all aquatic ecosystems except for deep marine systems (Ollis et al., 2013). In the South African National Water Act (Act No. 36 of 1998), a ‘wetland’ is more narrowly defined as:

> “land which is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface, or the land is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soil” (South African National Government, 1998: Chapter 1, Section1).

By this definition, a wetland is a unique type of aquatic ecosystem distinct from rivers and open water bodies (Ollis et al., 2013).

Wetland structure and function is driven primarily by its hydrology – the quantity, quality, distribution and timing of water flows (Macfarlane et al., 2008; Mitsch and Gosselink, 2015). Wetlands may receive water from rainwater, surface water, groundwater or a combination of these, and lose water through surface flows, groundwater flows, evaporation and evapotranspiration (Palmer et al., 2002). The specific features of individual wetlands are determined by these hydrological characteristics along with the interaction of multiple factors including the geology, soils, topography, climate and the influence of human activities (Palmer et al., 2002; Macfarlane et al., 2008). Variations in these factors result in a diversity of wetland types that differ in structure, characteristics and functioning. Further, wetland systems are naturally dynamic and influenced by external events such as major storms, long-term climate cycles and the geological evolution of the surrounding landscape and, depending on their specific characteristics, may respond differently to these external factors (Macfarlane et al., 2008).

2.2.1.2 Wetland functioning and integrity

Ecosystem integrity has evolved as a measure of ecosystem functioning and is used as a tool for ecosystem management (de Leo and Levin, 1997). It is an evaluative concept integrating several ecosystem attributes that together describe an ecosystem's ability to sustain its structure and function (Kay, 1991; Society of Ecological Restoration (SER), 2004). There is no single definition of ecosystem integrity; characteristics commonly associated with ecosystem integrity include function, structure, resilience and diversity (Okey, 1996; Turner et al., 2008). For example, "[e]cosystems with integrity are considered to be naturally self-organizing, dynamically stable, resilient and possess a diverse, native biota" (Okey, 1996:190). Ecological integrity is associated with a sense of being 'unimpaired' and 'whole'. An ecosystem that can preserve all its components (structure) and the functional relationships between them in the case of external disturbance has ecological integrity (Kay, 1991; Angermeier and Karr, 1994; de Leo and Levin, 1997). From an anthropocentric perspective, de Leo and Levin (1997) take this definition as reflecting the capability of an ecosystem to sustain 'services' of value to humans.

Ecological health is a related term, often used interchangeably with ecological integrity to describe the desired state of an ecosystem (Okey, 1996; SER, 2004). However, the two concepts are distinguished based on the nature of the desired ecosystem state. 'Integrity' is associated with an ecosystem state that resembles a 'reference'[17] ecosystem; while 'health' is associated with an ecosystem state displaying attributes within a certain 'normal' range (Kay, 1991; SER, 2004). In the field of wetland restoration, assessments and measures of ecological integrity and ecological health are used in planning and evaluating wetland interventions. In the practice of wetland rehabilitation in South Africa, it is considered insufficient to consider only the influence of rehabilitation interventions on the total size or area of a wetland; both the extent to which the health of the wetland and its potential to

[17] The SER International Primer on Ecological Restoration provides a discussion of a 'reference' state and how it can be defined (SER, 2004:8). In the context of wetland restoration, the reference state typically represents a point of advanced development on the intended trajectory of the restoration (SER, 2004).

provide ecosystem services are influenced by the rehabilitation and must be considered[18] (Kotze et al., 2009).

2.2.2 Wetlands and well-being

Humans and wetlands have a long association. Human-wetland interactions have taken various forms over time and include situations of humans living sustainably with wetlands, the Marsh Arabs of southern Iraq for example; of humans adapting to water-abundant surroundings (e.g., the Manobos people of the Agusan Marsh); of humans controlling and managing wetlands (e.g., ancient Babylonians and Egyptians developed water delivery systems involving wetlands); and humans fearing and destroying wetlands (prior to the 1970's, draining wetlands was an accepted practice across the world) (Mitsch and Gosselink, 2015; Denyer et al., 2018). More recently, wetlands are recognized globally for their part in maintaining the water cycle (Russi et al., 2013); as a source of biological diversity and genetic material (Ramsar, 2016); as sources, sinks and transformers of a number of chemical and biological materials (Mitsch and Gosselink, 2015); and their cultural significance (Denyer et al., 2018). In addition, they remain important for the well-being of many people who live near them (Kumar et al., 2011). Today, wetlands are revered, protected, restored and even created, but still drained, developed and destroyed.

In South Africa, the pattern is similar. Extensive investment in wetland rehabilitation has occurred (Phillips and Madlokazi, 2011; Kotze and Ellery, 2008) demonstrating the value of wetlands to people, yet the degradation and loss of wetlands has been significant (Kotze and Ellery, 2008) and both formal and informal development continues to encroach on, and impact, wetland ecosystems (Hay et al., 2014). Research on the contribution of wetlands to well-being across Africa is discussed by Hay et al. (2014). Findings demonstrate the range of benefits derived from wetlands and how the importance of these benefits is different to different people. It is apparent that not all wetlands contribute equally to the range and value of potential benefits associated with wetlands; some wetlands contribute relatively few benefits, while others are critical to the survival of many people. There appears to be a shift

[18]In planning and evaluating wetland rehabilitation in South Africa, tools and techniques are applied to assess wetland integrity and ecosystem service provision pre- and post- rehabilitation or for both the 'with' and 'without' rehabilitation scenarios (Kotze et al., 2009).

in the type of human-wetland interactions with a decrease in the ‘direct consumptive use’ of wetlands (e.g., reed harvesting), and an increase in non-consumptive use (e.g., recreation, direct use), water quality enhancement (indirect use) and particularly the importance of wetlands for water security (Hay et al., 2014). However, where there is consumptive use, this is often part of a vital livelihood strategy.

Wetlands play a fundamental role in sustaining human well-being and are increasingly thought of as socio-ecological systems emphasizing the inter-relations between wetlands and humans. Each wetland socio-ecological system is unique and to evaluate its specific contribution to human well-being necessitates an understanding of or familiarity with its particular context. The ecosystem services concept has emerged as a framing for articulating the relationship between ecosystems, their processes, and human well-being and is advocated as a framework to identify human-relevant wetland outcomes (Kotze et al., 2008; Turner et al., 2008; Russi et al., 2013; Mitsch and Gosselink, 2015; Ramsar, 2016). The ecosystem services approach and wetland ecosystem services are examined in the following sub-sections.

2.2.2.1 The ecosystem services framing

Well-functioning ecosystems are essential for human life and well-being. However, the specificities of this complex socio-ecological system are challenging to distil and articulate. The ecosystem services concept has been proposed as a way of conceptualizing the pathway between ecosystem structure and processes and human well-being. Ecosystem services are viewed as the ‘link’ between the properties and processes of ecosystems and human well-being. The concept was popularised by the 2005 Millennium Ecosystem Assessment (MA, 2005a) and has become a dominant conceptualization of the human-nature relationship (Muradian and Pascual, 2018). The MA (2005a:40) defined ecosystem services as “the benefits people obtain from ecosystems” and categorized provisioning, regulating, cultural and supporting services. The ecosystem services concept has evolved from this initial broad definition, and several definitions and classifications of ecosystem services have emerged (Nahlik et al., 2012).

One approach has been to distinguish between ecosystem processes and functions, ecosystem services, and human-well-being benefits (Boyd and Banzhaf, 2007; Fisher et al., 2009; Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011; Spangenberg et al., 2014a,b; Alexander et al., 2016; Turner et al., 2016). Ecosystem services are conceptualized as 'flowing' from natural capital (ecosystem structure and processes), and are transformed through interacting with other forms of capital into human benefits (Spangenberg et al., 2014a,b; Alexander et al., 2016; Turner et al., 2016). This conceptualisation is depicted in Figure 2.2. In this view, human well-being benefits derive from ecosystem services in combination with other forms of capitals (e.g., social, built and human capitals). In other words, ecosystem services and benefits, while linked, are not synonymous: ecosystem services lead to benefits, in combination with other inputs.

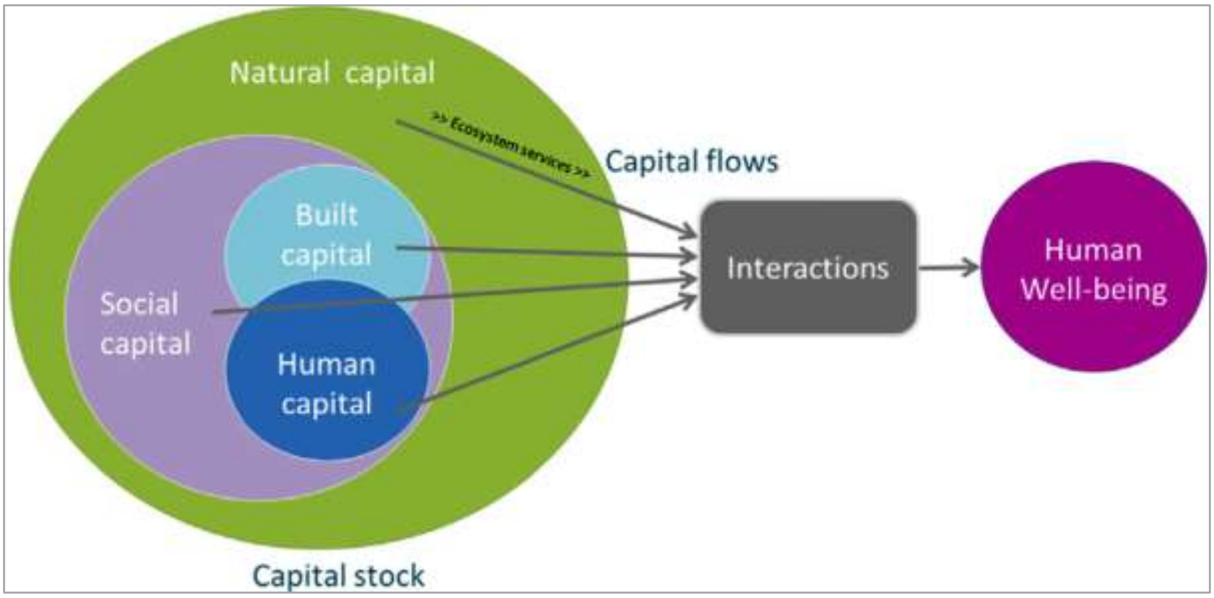


Figure 2.2: Schematic illustrating the interaction of capitals in the production of human well-being.

Source: Reproduced from Turner et al. (2016:193).

Haines-Young and Potschin (2010) and Potschin and Haines-Young (2011) emphasize the distinction between ecosystem services and benefits, placing ecosystem services between ecosystem 'functions' and 'benefits' in their 'cascade model' of the pathway from ecosystems to human well-being. Spangenberg et al. (2014a, b) modified the 'cascade model', introducing 'ecosystem service potentials' to emphasize that ecosystem structures, processes and functions exist regardless of human agency (or 'use'); ecosystem service potentials on the other hand, are the result of human agency in attributing 'use value' to ecosystem functions.

Drawing on the cascade model framing, a wetland may have the capacity to assimilate nitrogen (a function). If this function is recognized as 'useful' to humans (e.g., enhancing water quality) it 'becomes' an ecosystem service potential, but will only be 'mobilised' as an ecosystem service in the case of a human demand or need for the outcome of the function (e.g., in a catchment where nitrogen pollution affects humans). In other words, whether this function is regarded as a service or not depends upon human recognition of the function as a potential service and whether the outcome - 'water of an improved quality' - is considered a benefit by at least one person. The human well-being outcome, in combination with other 'capitals' (e.g., pipes and pumps for abstraction), is access to water of an improved quality (the benefit). This example highlights the importance of both the biophysical and social contexts in identifying the ecosystem services and benefits associated with a specific wetland system. Forming part of the social context are the various modes of governance influencing the system (Primmer et al., 2015). Viewed as a socio-ecological system, a holistic framing of the ecosystem services and benefits of a wetland requires consideration of the social, technical, economic, ecological and political dimensions and drivers that underpin the functioning of the system (Pollard et al., 2014).

Ecosystem structures and processes exist regardless of human beneficiaries (Boyd and Banzhaf, 2007; Potschin and Haines-Young, 2011; Spangenberg et al., 2014a,b), whereas the 'existence' of ecosystem services depends on whether the 'service' contributes to an outcome desired by at least one person (Alexander et al., 2016; Turner et al., 2016). Adopting an economic framing, an ecosystem service is a function of the biophysical conditions necessary to 'supply' the service and the 'demand' for the service or for the benefit to which the service contributes. From this perspective, ecosystem services are the outcomes of ecosystem structure and function that contribute to specific aspects of human well-being for which there is a demand. By this definition, the ecosystem services concept reflects an anthropocentric perspective of ecosystems, where natural capital is recognized for its contribution to human well-being.

2.2.2.2 Wetland ecosystem services

Wetland ecosystem services are commonly grouped into the categories of 'provisioning', 'regulating' 'cultural' and 'supporting' services, following the Millennium Ecosystem Assessment (MA, 2005a) typology. This classification is adopted in the Ramsar intergovernmental treaty on wetlands. Ecosystem service categories can be defined as follows (adapted from MA, 2005b; Haines-Young and Potschin, 2013; Turner et al., 2016):

a) Provisioning services are ecosystem services associated with nutritional, material and energetic outputs from living systems. Combined with built, human, and social capital, these services produce physical goods such as food and fibre. For example, crops cultivated in wetlands require at least farmers (human capital) and seedlings (manufactured capital) to benefit from the service.

b) Regulating (and maintenance) services refer to the ways ecosystems mediate or moderate (i.e., regulate) the ambient environment that affects human well-being. They include, for example, climate regulation (through carbon sequestration), water flow and quality regulation and pollination. The presence of human agency and additional 'capitals' transforms these ecosystem processes into ecosystem services and human well-being outcomes (benefits).

c) Cultural services are the processes and characteristics of ecosystems that, when combined with other capitals, produce outcomes and attributes of cultural significance. These can be viewed as the spiritual, symbolic, physical, and intellectual interactions of people with ecosystems contributing to, for example, cultural identity, recreation and education. There are challenges in defining this category of ecosystem services as (i) many of the benefits associated with cultural services are intangible and (ii) arguably all ecosystem services have a cultural dimension as it is the attribution of value (by people) to an ecosystem process or output that transforms it into an ecosystem service.

d) Supporting services are associated with ecosystem structure and function and are generally considered as underlying processes which support provisioning, regulating and cultural services, rather than being a division of ecosystem services. For this reason, there is controversy on their inclusion as a separate category of ecosystem services. The Common International Classification of Ecosystem Services (CICES, Haines-Young and Potschin, 2013) currently excludes 'supporting services' as a distinct category from an accounting perspective arguing that these 'services' underlie the other categories and therefore their exclusion is an attempt to avoid ‘double-counting’ in assessment and valuation exercises. In other classifications and uses (e.g., The Ramsar Convention - de Groot et al., 2006 and TEEB - Russi et al., 2013) the category is retained to highlight the importance of these underlying processes.

In South Africa, an ecosystem services framing has been incorporated into the design of the National Wetland Monitoring Programme (NWMP)[19] and the National Water Resource Classification System (WRCS). The ecosystem services approach is applied by wetland rehabilitation practitioners, including those involved in the National Working for Wetlands Programme[20]. The WET-EcoServices tool (Kotze et al., 2008) was designed as a method for the assessment of the provision of ecosystem services by wetlands tailored to the South African context. The tool was developed specifically to support practitioners of wetland rehabilitation to assess the ecosystem service outcomes of wetland rehabilitation projects and is used in the Working for Wetlands Programme to plan and evaluate wetland rehabilitation projects.

The WET-EcoServices tool version 2 (Kotze et al., 2020) differentiates 16 ecosystem services associated with South African inland, palustrine[21] wetlands, Table 2.2. The wetland services included in WET-EcoServices are those considered most important for South African wetlands which can be readily and rapidly described. The classification is not intended as an exhaustive list; other services, such as groundwater recharge and discharge may be relevant, but difficult to characterize at a rapid assessment level. While not strictly considered an ecosystem service, ‘biodiversity maintenance’ is included in the classification as it encompasses attributes widely acknowledged as having potentially high value to society (Kotze et al., 2008, 2020).

[19] The implementation of the NWMP falls under the National Aquatic Ecosystem Health Monitoring Programme (NAEHMP) of the Department of Water and Sanitation.

[20] The Working for Wetlands Programme is a national initiative that seeks to promote the protection, rehabilitation and wise use of wetlands in South Africa. The programme is part of a broader Expanded Pubic Works Programme to alleviate poverty and unemployment by providing work opportunities to the poor and unemployed.

[21] The term palustrine refers to “non-tidal wetlands dominated by emergent plants (e.g., reeds), shrubs or trees and includes a variety of systems commonly described as marsh, floodplain, vlei or seep” (Kotze et al., 2008:15).

Table 2.2: Ecosystem services associated with South African inland, palustrine wetlands included in the WET-EcoServices tool

Ecosystem service			**Description**
Regulating and supporting services		Flood attenuation	The spreading out and slowing down of floodwaters in the wetland/riparian area, thereby reducing the severity of floods downstream
Regulating and supporting services		Streamflow regulation	Sustaining streamflow during low-flow periods
Regulating and supporting services	Water quality enhancement services	Sediment trapping	The trapping and retention in the wetland/riparian area of sediment carried by runoff water
Regulating and supporting services	Water quality enhancement services	Phosphate assimilation	Removal by the wetland/riparian area of phosphates carried by runoff water, thereby enhancing water quality
Regulating and supporting services	Water quality enhancement services	Nitrate assimilation	Removal by the wetland/riparian area of nitrates carried by runoff water, thereby enhancing water quality
Regulating and supporting services	Water quality enhancement services	Toxicant & pathogen assimilation	Removal by the wetland/riparian area of toxicants (e.g., metals, biocides, salts) and pathogens carried by runoff water, thereby enhancing water quality
Regulating and supporting services	Water quality enhancement services	Erosion control	Controlling erosion within the wetland/riparian area, principally through the protection provided by vegetation
Regulating and supporting services		Carbon storage	The trapping of carbon by the wetland/riparian area, principally as soil organic matter
Biodiversity maintenance[1]			Through the provision of habitat and maintenance of natural processes by the wetland/ riparian area, a contribution is made to maintaining biodiversity
Provisioning services		Provision of water for human use	The provision of water which is taken directly from the wetland/riparian area for domestic, agricultural or other purposes
Provisioning services		Provision of harvestable resources	The provision of natural resources from the wetland/riparian area – including craft plants, fish, wood, etc.
Provisioning services		Food for livestock	The provision of grazing for livestock
Provisioning services		Provision of cultivated foods	The provision of cultivated foods from within the wetland/riparian area
Cultural services		Cultural and spiritual experience	Places of special cultural significance in the wetland/riparian area – e.g., for baptisms or gathering of culturally significant plants
Cultural services		Tourism and recreation	Sites of value for tourism and recreation in the wetland/riparian area, often associated with scenic beauty and abundant birdlife[2]
Cultural services		Education and research	Sites of value in the wetland/riparian area for education or research

Note: [1] It is recognized that biodiversity maintenance is not an ecosystem service in the strict sense (Liquete et al. 2016), and is framed in less anthropocentric terms than all of the other services, but it underpins many other services and is widely acknowledged as having high value to society broadly, even in the absence of any local or downstream beneficiaries.

[2] WET-EcoServices focuses on recreational services which are specifically nature-based, e.g., bird watching. It does not account specifically for recreational services from wetland/riparian areas that have been converted into sports grounds, children's playgrounds or other built infrastructure.

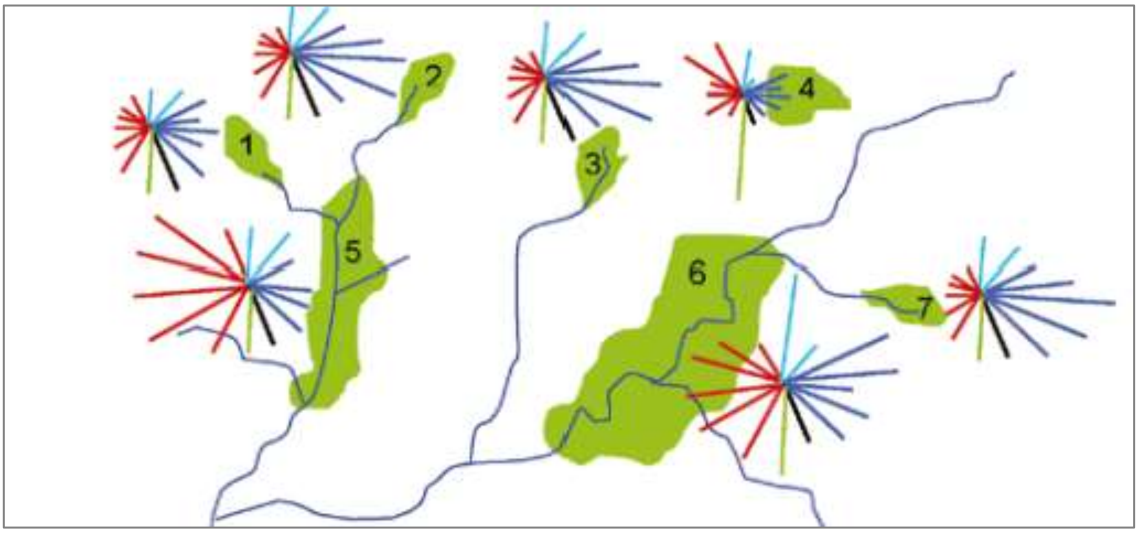
Source: Reproduced from Kotze et al. (2020:681).

Individual wetlands are not necessarily effective at providing all 16 ecosystem services, nor afforded the opportunity to do so. The potential of a wetland to supply a particular ecosystem service is determined by the biophysical and social context within which the wetland is situated. Biophysical characteristics include, for example, the hydrogeomorphic (HGM)[22] types present in the wetland, wetland slope and sinuosity, the prevailing climate and the size and integrity (condition) of the wetland. The WET-EcoServices tool identifies specific biophysical characteristics likely to influence each of the identified wetland ecosystem services, and it is these characteristics that are scored as an indication of the wetland's potential effectiveness (or capacity) to provide a specific ecosystem service.

The WET-EcoServices tool further distinguishes characteristics that give an indication of the opportunity afforded a wetland to provide a particular service. For example, a wetland may have the ability to assimilate nitrogen, but this function will only be 'mobilised' as an ecosystem service in the case of nitrogen pollution within the wetland's catchment. Additional characteristics reflecting 'opportunity' include land-use factors influencing runoff intensity, the extent of sediment sources within the catchment and the connectivity of the wetland to other natural areas in the landscape.

The provision of wetland services is a function of the specific biophysical and social characteristics of the system and differs between wetlands. Figure 2.3 demonstrates that the provision of ecosystem services differs across wetlands; the differences are in respect of differences in the biophysical capacity of the wetland to supply a service and differences in the social context in terms of the need or demand for the service. As noted by Mitch and Gosselink (2015: 527): "[p]erceived [ecosystem service] values arise from the functional ecological processes…, but are determined also by human perceptions, the location of a particular wetland, the human population pressures on it, and the extent of the resource".

[22] Hydrogeomorphic types are defined based on the geomorphic setting of the wetland in the landscape; the source of water to the wetland, how water flows through the wetland and how water exits the wetland (Kotze et al., 2008). There can be more than one type of HGM unit within a single wetland.



Note: The lines indicate the level of each of the services and illustrates how ecosystem service delivery differs between wetlands.

Figure 2.3: A spider diagram representation of seven hypothetical wetlands assessed in terms of their delivery of different ecosystem services.

Source: Reproduced from Kotze et al. (2008).

2.2.3 Wetland restoration

Despite their role in sustaining human well-being, wetlands have been subject to considerable degradation within South Africa, as in many countries. One of the globally accepted responses to this degradation is wetland restoration. Ecological restoration has an important role in conserving biodiversity and human well-being by addressing past (and on-going) degradation and, ideally, effecting a steady increase in the extent and integrity of ecosystems. As stated by the Society for Ecological Restoration (SER), it is vital, however, to acknowledge that "the promise of restoration should never be invoked as a justification for destroying or damaging existing ecosystems" (McDonald et al., 2016:8). Restoration should not be regarded as a substitute for protecting and sustainably managing undisturbed ecosystems as restoration science and practice cannot achieve a full recovery of ecosystem functioning and biodiversity (i.e., a return to a pre-disturbance state).

2.2.3.1 Restoration conventions, communities and programmes

Several global partnerships and organisations have emerged to call attention to the rate at which wetland ecosystems are being lost and to encourage their conservation and restoration. The Convention on Wetlands, termed the Ramsar Convention, is an intergovernmental treaty providing a framework for the conservation and wise use of wetlands (Ramsar, 2016). Since 1997, wetland restoration has featured in the Ramsar Strategic Plans and in 2002 Ramsar published the 'Principles and Guidelines for Wetland Restoration'. The vision of the fourth Ramsar Strategic Plan (2016–2024)[23], "Wetlands are conserved, wisely used, restored and their benefits are recognized and valued by all", includes wetland restoration as a goal and implies a need to articulate the benefits of wetland systems (Ramsar, 2016:16).

The Society for Ecological Restoration (SER) is a global non-profit organization of professionals with the goal of advancing the science, practice and policy of ecological restoration (SER, 2017). A global focus, and commitment to ecosystem restoration, is evident in the recent declaration of 'The Decade on Ecosystem Restoration, 2021 -2030' by the United Nations (UN) as one of their key environment programs (United Nations Environment Programme (UNEP) & Food and Agriculture Organization of the United Nations (FAO), 2019). The declaration emphasizes the importance of ecosystems, such as wetlands, for sustainable development, poverty alleviation and improved human well-being. The principles for ecosystem restoration developed as a guide to the UN Decade of Restoration emphasize the need for monitoring and evaluation throughout the lifetime of a restoration intervention or project and the importance of the local ecological, cultural and socio-economic contexts and the integration of different types of knowledge in restoration design and implementation (FAO et al., 2021).

In South Africa, the Working for Wetlands Programme, a national government initiative, seeks to promote the protection, rehabilitation and wise use of wetlands. The programme was initiated in 2002 within the broader Expanded Public Works Programme and uses wetland rehabilitation as a vehicle for both poverty alleviation and sustainable wetland management. The programme follows an approach centred on cooperative governance and the creation of partnerships between landowners, communities, civil society and the private sector.

[23] The Ramsar Strategic Plan 2016–2024 was adopted at the twelfth meeting of the Conference of the Parties to the Convention on Wetlands of International Importance (Punta del Este, Uruguay, 2015).

Monitoring and evaluation within the Working for Wetlands Programme has largely focused on programme outputs as reflected in its performance indicators (e.g., number of person days, number of jobs created, number of wetlands rehabilitated). Programme achievements since 2004 include an investment of R1.3 billion in the rehabilitation of more than 1 500 wetlands, creating approximately 37 000 jobs (Department of Environment Forestry and Fisheries (DEFF), 2021). Stimulated by the large-scale investment of public funds in wetland rehabilitation, an eleven volume series of reports, manuals and guidelines for wetland assessment and rehabilitation in South Africa (The Wetland Management Series) was published by the Water Research Commission (Dada et al., 2007). The set of guidelines has formed the scientific and technical foundation for the programme's wetland interventions.

In addition to the Working for Wetlands Programme, wetland intervention projects are undertaken by organizations and individuals throughout South Africa, often in compliance with authorization conditions in terms of the National Environmental Management Act No 107 of 1998 (NEMA) and National Water Act No. 36 of 1998 (NWA) as a means of mitigating the impacts associated with a proposed activity. Monitoring and evaluation of these wetland intervention projects tends to be limited and inconsistent, with various techniques being used to measure the mitigation outcomes.

As part of its commitment to the United Nations Convention to Combat Desertification (UNCCD) and linked to Sustainable Development Goal (SDG) 15, specifically Target 15.3, South Africa has set targets towards achieving land degradation neutrality by 2030. One of the targets is to rehabilitate 61 900 ha of wetlands by 2030 as compared to the baseline extent in 2015 (DEA, 2018). Such targets, along with environmental legislation regulating the impact of land-use change and impacts affecting wetland systems, indicate that investment in the rehabilitation of wetlands in South Africa will continue and is likely to increase.

2.2.3.2 Terms and definitions

In lay terms 'restoration' is associated with the return of something to its original state, and in the past, ecological restoration was associated with attempts to return an ecosystem to (some) historical state (Higgs et al., 2014). However, this conventional notion of restoration no longer fits the suit of interventions undertaken in practice (Allison, 2007) nor recognizes the limitations of restoration to achieve a pre-disturbance state. In the wetland context, the field of practice has expanded to encompass a broad range of activities to repair degraded wetlands, offset damage to ecosystems and to provide or enhance specific ecosystem services (construct or design for a desired purpose). A large proportion of contemporary ecological 'restoration' activities aim for a desired or novel state rather than a return to some 'original' or historical state (Rohwer and Marris, 2016).

A widely cited definition of ecological restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed" (SER, 2004:3). This definition associates ecological restoration with interventions that aim to return, or direct, a degraded ecosystem back to its historical, 'natural' trajectory (i.e., not to a fixed state) in contrast to imposing a novel direction or state (McDonald et al., 2016). In this way, 'restoration' is broadly distinguished from 'rehabilitation' which is associated with reinstating a level of ecosystem functionality and ecosystem service delivery (McDonald et al., 2016). However, rather than imposing a strict and exclusionary division between interventions that aim to improve environmental conditions, the SER has developed a 'restorative continuum' (Figure 2.4), which conceptualizes interventions as degrees of restorative activity (McDonald et al., 2016; McDonald and Dixon, 2018; Gann et al., 2019). Restorative activities, including those that reduce negative impacts or effect ecosystem repair, are considered complementary to ecological restoration by improving conditions for broad scale ecosystem recovery.

The Ramsar Convention on Wetlands does not clearly separate 'restoration' and 'rehabilitation', noting that the terms are often used interchangeably, and uses 'restoration' in a broad sense to include interventions that aim to return a wetland to pre-disturbance conditions and interventions that aim to improve wetland functioning without necessarily returning to it to a previous state (Ramsar, 2010). In the South African wetland science and management context, interventions in wetland ecosystems are generally referred to as 'rehabilitation' in recognition of the uncertainty in attaining some former (prior to anthropogenic disturbance) state and, increasingly, the case of designing wetland systems towards a desired state (e.g., to promote the supply of a particular ecosystem service). Rehabilitation interventions are not expected to return a wetland to a former state. Wetland rehabilitation is considered a process of assisting in "the recovery of a degraded wetland's health and ecosystem service delivery by reinstating the natural ecological driving forces or, halting the decline in health of a wetland that is in the process of degrading, so as to maintain its health and ecosystem service delivery” (Kotze et al., 2008:14).

Figure 2.4: The restorative continuum adopted by the Society of Ecological Restoration. Source: Reproduced from Gann et al. (2019:S22).

This definition emphasises that wetland rehabilitation interventions can be intended to improve or increase the potential of a wetland to supply ecosystem services, or to prevent a decline or loss in potential wetland services. The implication for evaluating intervention outcomes is that the ‘with’ and ‘without’ rehabilitation scenarios must be compared rather than the ‘with rehabilitation’ and the ‘pre-rehabilitation’ cases. An outcome of the rehabilitation of ‘no change’ (i.e., no loss) in the potential of the wetland to supply ecosystem systems services may still be a positive outcome in the case of continued degradation without the intervention leading to a decline in ecosystem service supply potential. Wetlands are naturally dynamic systems, meaning that it can be challenging to formulate the ‘with’ and ‘without’ rehabilitation scenarios, that is, to predict precisely how the wetland will respond to a particular intervention or the outcomes of continued degradation without the intervention (Dada et al., 2007).

2.2.3.3 The restoration process

A wetland restoration project refers to a set of interventions designed to achieve a recovery in the condition of a degraded wetland or halt a decline in the condition of a wetland that is in the process of degrading. This recovery is not immediate and can continue for many years following the initial interventions (Streever, 1997; Moreno-Mateos et al., 2012). A holistic approach to wetland restoration takes a long-term view that extends beyond the initial repair interventions (e.g., construction works) to include maintenance and aftercare activities required to support the initial intervention(s) (Ramsar, 2002; Kotze et al., 2008). Along with a transparent and structured planning stage, long-term monitoring, evaluation and maintenance are considered an essential part of the restoration process (Kotze et al., 2008).

Wetland restorations are planned with the intention of achieving one or more objectives. In the South African wetland rehabilitation field, outputs are distinguished from outcomes, particularly in the context of evaluation. Outputs describe specific interventions (e.g., a gabion weir to control erosion); whereas outcomes are the results the interventions were designed to achieve (e.g., major gully erosion halted in the wetland, thereby securing the integrity of an intact portion of the wetland under threat from erosion) (Cowden and Kotze, 2008; Walters et al., 2019). Outcomes relate to the physical, chemical and ecological response of the wetland to the restoration interventions; this response takes time and outcomes are generally not achieved immediately following implementation.

Wetland restoration projects can have both output objectives and outcome objectives. Output objectives are used to evaluate restoration implementation and are usually achieved during, or within a year of, implementation and are relatively easy to measure (Rutherfurd et al., 2000). In the context of the South African National Working for Wetlands Programme, social outputs are an important overall programme objective and relate to job creation and the number of person days generated by the rehabilitation project. These social outputs are not specific to the response of the wetland to the rehabilitation, but rather are linked to implementation effort. Outcome objectives, in contrast, are specific to the response of the wetland to the rehabilitation efforts and may be hydro-geochemical (e.g., raising the water table), ecological (e.g., creating habitat for a specific species) and / or social (e.g., enhancing the potential of the wetland to provide ecosystem services) and are typically attained (or observed) over a much longer timeframe and can be more difficult to measure (Rutherfurd et al., 2000). Outcome objectives and measures provide the basis for evaluating the human well-being benefits specific to the wetland restoration (Clewell et al., 2005).

The various activities of the wetland restoration process can be grouped as planning and design; implementation (establishment and construction); monitoring, recording and evaluation; and post-implementation maintenance and management (Rutherfurd et al., 2000; Ramsar, 2002; McDonald et al., 2016). Resources (effort and materials) are required for each of these activities which means that the cost of a wetland restoration project extends beyond the costs of implementation. These additional costs are not always included in restoration budgets, nor recorded and / or clearly reported (Robbins and Daniels, 2012; de Groot et al., 2013; Meli et al., 2014).

Broadly, restoration planning and design entails activities related to identifying the restoration goals and targets through stakeholder engagement; undertaking ecological and contextual assessments to set the baseline and identify activities that degrade or threaten the health of the wetland; conducting risk assessments; and developing an appropriate restoration strategy and detailed designs for physical infrastructure (Rutherfurd et al., 2000; Ramsar, 2002; McDonald et al., 2016). The manual of Kotze et al. (2009) provides a framework for wetland rehabilitation planning in the South African context. The framework highlights that restoration planners may need to undertake environmental impact assessments and apply for legal authorization for the specific interventions which can add significantly to wetland restoration planning costs. Rutherfurd et al. (2000), for example, noted that planning and design costs could comprise 20 percent of the overall restoration cost and recommended allowing for five to 10 percent of the project cost for planning and design.

The implementation phase involves undertaking the activities set out in the restoration plan towards achieving the project objectives (Clewell et al., 2005). Wetland restoration activities can take the form of interventions in the biophysical system and / or the social system and can include for example:

- Civil works (e.g., concrete, earth or gabion structures) to arrest erosion, trap sediment and re-saturate drained wetland areas;

- Using structures and landscaping to reinstate diminished flood mitigation and water quality enhancement functions;
- Eradicating invasive alien plants and re-vegetation of the wetland area;
- Addressing offsite causes of degradation, such as inappropriate agricultural practices;
- Raising awareness of the importance of, and management / wise use of, wetlands;
- Providing technical skills; and
- Developing management and restoration plans (drawing from Dada et al., 2007:8).

The costs associated with implementation depend on the type and number of interventions and particularly the extent of earthworks and engineered structures required (Sparks et al., 2013). Generally, construction costs are the largest cost over the restoration project, unless the restoration process includes the purchase of land (Rutherfurd et al., 2000; Sparks et al., 2013; Peh et al., 2014).

Ecological restoration is characterized by complexity and uncertainty and the effectiveness of restoration interventions is considered highly variable (Suding, 2011; Iftekhar et al., 2017). Monitoring, evaluation and adaptive management assist in dealing with this inherent uncertainty and are considered critical components of ecosystem restoration projects (SER, 2004, Rutherfurd et al., 2000; Ramsar, 2002; Clarkson, 2012; McDonald et al., 2016). Monitoring is the systematic collection of data based on observations and measurement of change, guided by a pre-defined question or objective, leading to new information (Larson et al., 2013). Monitoring goes beyond data collection and includes analysis to provide insights on wetland functioning and how this functioning changes with restoration (Clarkson, 2012)[24]. Evaluation entails the comparison of project outcomes to project goals and an interrogation of any divergence and is generally based on the results of monitoring (Rutherfurd et al., 2000; Hermans et al., 2012). Adaptive management is a management approach whereby future management is continuously adjusted in response to new information provided through monitoring and evaluation (Larson et al., 2013).

There is a continuum of complexity, need for expertise and cost across different levels of monitoring and evaluation. For example, changes in vegetation composition can be measured

[24] Similarly, Rutherfurd et al. (2000) view monitoring as a form of continuous evaluation.

adequately through a before and after restoration sampling. However, to be confident that the effects observed are a consequence of the restoration, replicated sampling coupled to a control site may be required. The former may be adequate for management purposes, but the latter would be required from a “scientific certainty” perspective. The extent of investment in monitoring and evaluation depends on the purpose of the information, the desired confidence in the findings and the type of data required.

While wetland restorations should be designed to be as self-maintaining as possible, in most cases some form of maintenance is necessary to support the initial interventions (Rutherfurd et al., 2000; Ramsar, 2002; Kotze et al., 2008). The degree of maintenance needed is influenced by the biophysical context (e.g., flood events, type and extent of built structures) and the social context (e.g., surrounding land use such as livestock grazing, the risk of vandalism). Most engineered structures have a finite design life, accordingly, the restoration itself may have a finite life; in other cases, the structures may have positive effects on the system that will continue well beyond the life of the structures. The wetland vegetation will generally require some form of management for example the removal of biomass and the control of alien invasive plant species. Maintenance and aftercare activities may be required on a regular basis (e.g., vegetation management) as well as on an ‘as needed’ basis (e.g., structural repairs). If issues requiring maintenance are not attended to timeously, they may develop into major issues requiring greater investment or reducing the ‘life of the restoration’. Maintenance needs may vary from year to year, and be challenging to predict, and, therefore, routine monitoring or observation may be required.

The costs associated with the various phases of wetland restoration from planning to post-implementation maintenance are broadly categorised as investment costs (planning, including initial baseline assessments and environmental authorisation applications[25], and implementation) and operating and maintenance costs (monitoring and evaluation, maintenance and aftercare) (e.g., Prato and Hay, 2006; Alfranca et al., 2011; Roebeling et al.,

[25]Wetland restoration can be environmentally disruptive and may require legal permits or authorisation (Armstrong, 2008).

2011). Land acquisition (Roebeling et al., 2011), infrastructure depreciation[26] (Alfranca et al., 2011); education and awareness (Prato and Hay, 2006); additional infrastructure and facilities associated with mobilizing desired ecosystem services (e.g., a bird hide for tourism; excavations and structures to support water treatment services); and supporting infrastructure (e.g., a crossing point over a channel for community and livestock use) are additional cost elements that may be relevant depending on the specific context.

2.2.4 Section conclusion

Wetland restoration is a globally accepted response to the degradation of wetlands and aims to steadily increase the extent and integrity or health of wetland ecosystems. In this research, 'wetland restoration' is used in a broad sense to describe interventions that aim to assist in the recovery of a degraded wetland or to halt a decline in health of a wetland that is in the process of degrading[27]. In concluding this section, several points particularly relevant to the economic evaluation of wetland restoration are summarized.

- Evaluating wetland restoration from an economic value perspective requires that the contribution of the restoration to human well-being be identified and measured.
- The biophysical outcomes of the restoration provide the basis for evaluating the human well-being benefits specific to the wetland restoration.
- Conceptually, the biophysical outcomes of wetland restoration can be linked to human well-being through the ecosystem services framing[28]. The approach is compatible with an economic value framing (anthropocentric, utilitarianism), but not necessarily with other value theories.
- Human well-being benefits derive from wetland ecosystem services in combination with other forms of capitals (e.g., human resources, built capital).
- Wetlands have the potential to supply a range of ecosystem services; this potential depends on the specific biophysical characteristics and integrity of the wetland and

[26]Infrastructure depreciation costs are not generally considered in the context of public works, but may be included in the case of private investment in restoration (Alfranca et al., 2011).

[27] This definition aligns with the definition of wetland rehabilitation adopted in South African Working for Wetlands Programme.

[28] Specifically the conceptualizations of the ecosystem service concept by Potschin and Haines-Young (2011), Spangenberg et al. (2014a,b), Alexander et al. (2016) and Turner et al. (2016).

the catchment. This means that the potential to provide ecosystem services differs across individual wetlands.

- Wetland restoration can enhance, maintain or even reduce individual ecosystem service potentials of the wetland – this influence can differ across ecosystem services. Further, restoration activities can be designed to enhance a particular ecosystem service.
- While a wetland may have the potential to provide a range of ecosystem services, ‘mobilisation’ of one ecosystem service may compromise the ‘mobilisation’ of another. For example, making use of a wetland to improve water quality may mean forgoing the wetland as a source of water for livestock.
- Both the biophysical and social contexts within which the wetland restoration takes place are essential to identifying the restoration outcomes, ecosystem services and benefits of the restoration.
- The process of wetland restoration entails planning, implementation, monitoring and evaluation, and post-implementation maintenance. There are costs associated with each of these stages, which are not always included in restoration project budgets, nor recorded and / or clearly reported.

2.3 ECONOMIC ECOSYSTEM VALUATION – METHODS AND CHALLENGES

"Each of us has a way of making sense of the world"

Heinzen (2004:iii)

Wetland restoration is a choice between competing alternatives in terms of both multiple land-use options and alternative uses of the resources required to implement restoration. Any choice between alternatives implies some form of comparison to determine which alternative is preferred. Conceptually, alternatives are evaluated by identifying the favourable and adverse outcomes of each alternative, the 'benefits' and 'costs', and comparing them. However, argued Freeman (2003), the terms 'benefits' and 'costs' are meaningless unless set against a specific goal or objective. It is the underlying goal which defines whether an outcome is favourable or not. Economic valuation is intended as a means of expressing the costs and benefits of alternative actions in a common measure to facilitate comparison (evaluation) and decision-making. In so doing, it adopts the specific goal of economic efficiency and human welfare consequences in defining 'costs' and 'benefits'. Viewed as a way to measure and compare the benefits and costs of wetland restoration to competing alternatives, economic valuation, and CBA, is regarded as a useful tool to aid global wetland management (Barbier et al., 1997; Woodward and Wui, 2001; Braat and de Groot, 2012; Chaikumbung et al., 2016).

Valuing wetland restoration involves determining the effect of the restoration efforts on the wetland and measuring and valuing the resulting outcomes. It is about the difference in benefit flows with the restoration, rather than a summation of total wetland benefits, and implies a comparison of the 'with' and 'without' restoration states (Bockstael et al. 2000; Pagiola et al. 2004; Heal et al., 2005; Pendleton and Baldera, 2010). The conventional cost-benefit evaluation of wetland restoration involves assigning an economic value, measured in a monetary metric, to the difference in benefit flows 'with' and 'without' restoration and comparing these to the costs, measured in a monetary metric, of the restoration. This type of evaluation is typically undertaken to justify capital expenditure on wetland restoration, either *a priori* to demonstrate economic rationality or *a posteriori* to motivate for further investment or to inform a specific technical or policy decision.

In the first part of this section economic ecosystem valuation is considered and the typical rationale for its application is introduced. An overview of economic ecosystem valuation methods follows. The final section explores a number of critical perspectives on, and challenges to, economic ecosystem valuation.

2.3.1 Economic ecosystem valuation

2.3.1.1 Valuation – a process of value construction

Valuation is an action. It is a process of attaching a particular importance to an object or phenomena. In the context of ecological restoration, valuation is the process of attaching a type and degree of importance (value) to the outcomes of the restoration. Values are not 'given' or fixed and merely revealed by valuation; they are shaped by the economic, social, ecological and political interactions of everyday life, and are assembled through the valuation process (Henrich et al, 2001; Klamer, 2003; Martín-López et al., 2014). The method of valuation shapes the nature of this process (Vatn and Bromley, 1994). Valuation methods emerge from a particular understanding of what values are, or should be, and how they can be known; they are designed according to the pre-analytic conceptions of the designers (Brondízio et al., 2010), and each employs its own 'language of valuation' (Martinez-Alier, 2008; Spangenberg and Settele, 2016). Values and valuation processes are not ideologically neutral.

Vatn (2005) described valuation methods as value articulating institutions which define "a set of rules concerning the valuing process" (Vatn, 2005:301). These 'rules' are the ontological and epistemological judgements that underpin the valuation approach and determine the value system applied. As such, valuation methods impose a specific way of thinking on the valuation process, thereby shaping and defining the values that they 'elicit' (Vatn and Bromley, 1994; Martín-López et al., 2014). In this way, "valuation processes can act as vehicles that articulate particular notions of property and ownership, rationalities, and ways to relate with the environment that are specific to particular societies" (Gomez-Baggethun et al., 2010:1215). In the words of Gomez-Baggethun et al. (2014:20), "[v]aluation methods and associated rationalities are frames invoked in the process of expressing values that regulate and influence which values come forward, which are excluded, and what sort of conclusions can be reached". It follows that economic valuation methods too, impose a specific framing and rationality on the formation and development of values associated with ecosystems and ecological restoration.

2.3.1.2 Economic ecosystem valuation

Economic ecosystem valuation is the process of attaching economic value to the contributions of ecosystems and biodiversity to people. It is an attempt to provide an empirical account of the value of these contributions or of the benefits and costs of actions that modify the flow of ecosystem and biodiversity contributions to people (National Research Council, 1999). By definition, economic value is utilitarian[29] and anthropocentric and reflects a very particular view of value based on the satisfaction of human preferences under a specific model of human behaviour (Young, 1996). Within a utilitarian framing, value is embedded in the goals to which individuals and society aspires (Costanza, 2000; Liu et al., 2010). The economic evaluation of wetland restoration then, is the process of assessing the contribution of the restoration to the goal of economic efficiency – that is, the allocation of resources in such a way as to achieve a situation where it is not possible to improve the welfare of an individual without reducing the welfare of another. The foundation of the economic evaluation of wetland restoration is welfare-change measurement: the benefit (or cost) of the restoration is measured as the welfare change that it generates.

Conceptually, the 'accepted' measure of this welfare change is an individual's willingness to pay (WTP) for a welfare gain (benefit) or to avoid a welfare loss (cost) or their willingness to accept compensation (WTA) to forgo a benefit or incur a cost (Pearce and Secommbe-Hett, 2000). These welfare changes are aggregated across (all 'affected') individuals as a measure of social welfare change. Where ecosystem products and services are traded in a market, WTP and WTA measures are reflected in market prices. However, many of the functions and benefits of ecosystems, including wetlands, do not 'appear' in markets (Turner et al., 2000).

[29] Here, utilitarian in the sense of individual preference satisfaction, however, other interpretations of utility and how it can be measured exist.

At one extreme are ecosystem products used and traded directly (e.g., certain kinds of wetland reeds); at the other extreme are non-rival and non-excludable public services (e.g., the cultural identity associated with a particular wetland or the contribution of wetland habitat to carbon storage and climate stability) (Farley, 2008). Arguably, both contributions have economic value in the sense of individual preference satisfaction. However, ecosystem attributes and functions not traded directly in markets remain 'unpriced'. Economic valuation methods have evolved over time in an attempt to assign economic value to the contributions of ecosystems to human well-being; one intention being to incorporate these contributions into conventional decision-making processes. The expectation is that utilitarian arguments about ecosystem benefits – expressed as economic values (often conflated with monetary measures) – will reduce ecosystem and biodiversity degradation and provide support for ecosystem restoration.

2.3.1.3 The rationale for economic ecosystem valuation

It is widely accepted that the vital role of ecosystems and biodiversity in maintaining human well-being has, until recent decades, largely been ignored in land-use decisions[30] (Barbier et al., 1997; Costanza et al., 1997a; Heal et al., 2005) leading to ecosystem degradation and biodiversity loss (Birol et al., 2006). From a neoclassical economics perspective, this is attributed to 'market failure' whereby many of the contributions of ecosystems, and the consequences of ecosystem degradation, remain 'unpriced' and left out of conventional economic CBA processes. CBA, based on the economic efficiency criterion, is considered by many as an important source of information in decision- and policy-making (Turner et al., 2000; Heal et al., 2005; Brander et al., 2012) and is regarded by its supporters as a "powerful, widely used and relatively easy tool for deciding whether to make a change" (Naess, 2006a:34).

The fundamental assumption of the neoclassical environmental economics perspective is that environmental degradation results from market failure, specifically the absence of markets for ecosystem goods and services. Market failure is primarily attributed to missing property rights; without property rights there is no market, no market exchange and thus no economic

[30] That is, in the Western sense of land-use and development.

(exchange) value (Papandreou, 2003). In the absence of measured economic exchange values the market 'fails' to achieve an efficient allocation of resources. The public benefits of wetlands (e.g., water quality enhancement, streamflow regulation, climate stability), for example, are not typically reflected in markets, whereas the benefits, often private benefits, associated with alternative uses of the land (e.g., industry, agriculture, housing) are more likely to be reflected in markets (Gutrich and Hitzhusen, 2004). Historically, the economic justification for the conversion of wetlands to alternative land-uses has outweighed the justification for their preservation.

From a neoclassical economics perspective 'getting the prices right' – or assigning economic value – so that the benefits of ecosystems, and the costs of ecosystem degradation, are included in CBA is regarded as a solution (Farley, 2008; Centemeri, 2009). Economic ecosystem valuation is perceived as a way to redress the neglect of ecosystems in land-use decision-making (Chee, 2004). This view assumes that for ecosystems, their integrity, and biodiversity to be considered in decision-making processes, their contribution to human well-being must be ascertained and articulated in the 'language of economics' (Turner et al., 2000; de Groot et al., 2012), which involves assigning economic values to ecosystem and biodiversity contributions (Barbier et al., 1997; Carpenter et al., 2006; Farley, 2008). Economic value estimates expressed in monetary units are viewed as the language of business and policy-making (e.g., Constanza et al., 1997; Balmford et al., 2002) as they translate nature's contributions into terms familiar to decision-makers and the public.

Economic ecosystem valuation is used to assign economic values to the benefits and costs of changes in ecosystem condition in a common metric to facilitate comparison with the costs and benefits of alternative actions (Pritchard et al., 2000; Pagiola et al., 2004; EFTEC, 2006). This is the general intention in the economic evaluation of wetland restoration actions. As argued by Heal et al. (2005:2):

> "choices between the conservation and restoration of some ecosystems and the continuation and expansion of human activities in others have to be made … [in] making these choices, the economic values of these ecosystem goods and services to society have to be known, so that they can be compared with the economic values of activities

> that may compromise them and improvements to one ecosystem can be compared to those in another".

In this type of application:

> "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).

In this way, economic valuation information is seen as supporting improved decision- and policy-making (from the perspective of achieving economic efficiency) in land-use and environmental management (Carpenter et al., 2006; de Groot et al., 2010; Brouwer and Georgiou, 2012).

In the context of wetland management, Birol et al. (2006), for example, asserted that capturing the total economic value (based on the TEV framing) of the wetland provided policy-makers with the necessary information to develop a strategy for the efficient management of the wetland and concluded that "[c]apturing the TEV of water resources is an integral part in the design of economic incentives and institutional arrangements that can ensure their sustainable, efficient and equitable allocation" (Birol et al., 2006:119). Weber and Stewart (2009) concluded that their economic valuation of the benefits of riverine restoration provided economic justification for riparian restoration and management leading them to suggest that "the inclusion of numerical estimates of benefits, based on carefully collected and professionally analyzed public preference data, offers a significant improvement to public land management decision-making" (Weber and Stewart, 2009:770). From a meta-analysis of the costs and benefits of ecological restoration projects, de Groot et al. (2013), while acknowledging the complexity of 'pricing' ecosystem services, emphasised that economic valuation information has a significant influence on planning and decision-making and concluded that cost-benefit analyses can be useful in justifying ecological restoration. Similarly, Iftekhar et al. (2017:267) advocated "the sound application of economic principles and tools" in restoration planning and implementation, noting the risk of undervaluing restoration in investment decisions if the full range of benefits and costs are not valued.

Properly used, proponents believe that economic ecosystem valuation can provide valuable insights into land-use and ecosystem management decisions (e.g., Arrow et al., 2004; Braat and de Groot, 2012). For example, The Economics of Ecosystems and Biodiversity (TEEB) is a major international initiative to mainstream the value of ecosystem services and biodiversity into governance and decision-making by adopting an economic framing and language (Kumar et al., 2010). One of the aims of the initiative is to highlight the economic significance of ecosystems and biodiversity, with an emphasis on "economic, notably monetary, effects" (de Groot et al., 2010:5). The stated aim of TEEB for Water and Wetlands is to "show how recognizing, demonstrating, and capturing the values of ecosystem services related to water and wetlands can lead to better informed, more efficient, and fairer decision making" (Russi et al., 2013:1).

However, even supporters, including many economists, recognize the complexity and limitations of economic ecosystem valuation and argue that economic valuation methods cannot, and are not intended to, reflect all the values of ecosystems and biodiversity, nor are they the only way to assess the contributions of ecosystems and biodiversity (e.g., Hanemann, 1994; Bockstael et al., 2000; Pagiola et al., 2004; Heal et al., 2005; Pascual et al., 2010). Specifically with respect to monetary-based valuations, the TEEB (2010) study concluded that:

> "It should become clear that techniques to place a monetary value on biodiversity and ecosystem services are fraught with complications, only some of which currently can be addressed…. [v]aluation exercises can still provide information that is an indispensable component of environmental policy… policy-makers should interpret and utilize the valuable information provided by these techniques while acknowledging the limitations of this information" (Pascual et al., 2010:66).

Economic ecosystem valuation is not unilaterally accepted as the best means, or even as appropriate, for informing land-use decision-making and ecosystem governance. A further complication in this respect is the conflation of economic valuation with monetary value measurements and a lack of attention alternative metrics for value articulation. Several economic valuation challenges and critical reflections are considered in section 2.3.3. Since a number of these relate to methodological uncertainties, economic valuation methods applied to assign economic value to ecosystem and biodiversity contributions are first discussed.

2.3.2 Economic valuation methods

The economic valuation of ecosystem benefits is approached either through primary valuation or by secondary valuation. Primary valuation involves the direct application of economic valuation methods to the ecosystem benefits under consideration. Secondary valuation involves the 'transfer' of value information from primary valuation studies to a new context or site (Brouwer et al., 2013) and is generally referred to as value transfer. The two approaches are often combined in a single assessment.

Several methods are available for primary economic valuation and are broadly categorised as market-based, non-market-based and cost-based methods[31]. Market and non-market-based methods are grounded in economic welfare theory and aim to measure individuals' willingness to pay (WTP), or willingness to accept compensation (WTA), for changes in their welfare associated with alternative ecosystem states. Cost-based methods, on the other hand, do not measure WTP or WTA directly and are used as a proxy of economic value (Heal et al., 2005).

Economic valuation methods attempt to infer or elicit human preferences regarding changes in ecosystem state. Most commonly, these preferences are expressed using a monetary metric; however, alternative metrics have been suggested. For example, Fisher et al. (2008) propose that an index of vulnerability, happiness or number of lives saved could be used instead of monetary units. The overall intention is to express values in a common metric to facilitate comparison and monetary units are widely recognized and readily understood by decision-makers and the public. Using a monetary metric to express ecosystem benefit values does not mean that only 'money-generating' benefits are being considered (Pagiola et al., 2004).

There is a well-developed literature on economic ecosystem valuation methods. General texts, for example, Champ et al. (2003), Pagiola et al. (2004) and Hanley and Barbier (2009)

[31] The categorisation used here is widely accepted, however, various other groupings and terminologies of economic ecosystem valuation methods are used (Boyle, 2003; de Groot et al., 2006).

provide detail on the theory of valuation techniques, Heal et al. (2005) focus specifically on the valuation of aquatic ecosystem services, and de Groot et al. (2006), Turner et al. (2008) and Turpie and Kleynhans (2010) concentrate on wetland benefit valuation. The aim of this section is to introduce the range of methods drawing attention to their use in both the South African and wetland ecosystem context. A challenge common across the methods in the specific application of valuing rehabilitation outcomes is the need to define the relationships (causality) between the rehabilitation effort, the resulting difference in wetland condition between the 'with' and 'without' rehabilitation cases and the benefit outcome(s), which may not be well understood, predictable or quantifiable.

2.3.2.1 Market-based methods

Market-based valuation methods make use of existing market transactions to derive the value of ecosystem benefits (Turner et al., 2008). Two approaches are noted; the market price method and the production function method. For the case of ecosystem goods or services that are traded, wetland reeds or wetland carbon storage, for example, the market price of a unit of the traded good or service can be used to value the benefit associated with the ecosystem, or the benefit resulting from a change in the condition of the ecosystem. Standard consumer and producer surplus theory applies (Turner et al., 2008). Where market prices are distorted (e.g., as a result of subsidies) prices may need to be adjusted and 'shadow prices' estimated to reflect the true economic value of the benefit (Barbier et al., 1997).

In South Africa, the market price method has been used to value an increased supply of marketable goods and services from alien plant clearing and restoration of indigenous fynbos (Fourie et al., 2013). The market price approach is regarded as a robust method of valuation as estimates are based on observed data of actual consumer preferences. The method is limited to ecosystem benefits that are 'private goods' and traded directly in a market and the approach is not suitable in the case of large-scale changes likely to affect the supply or demand for a good or service. There is a risk of overstating the ecosystem benefit if the costs of other inputs used to bring the ecosystem good or service to the market are not considered.

The production function method treats an ecosystem attribute or function as an 'input' into the production of a market good (Barbier, 2007). A production function is specified, which relates the production of 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). In this way, the value of the ecosystem benefit is derived from the change in the production of the marketed good or service as a result of changes in ecosystem condition. The method has been used in South Africa to value the consequences of salinization of irrigation water for commercial irrigated agriculture (de Lange et al., 2012) and to investigate the relationship between elements of estuarine biodiversity and the recreational fishery economy (Crafford and Hassan, 2014). The method is generally regarded as robust and conceptually straightforward; however, it can be practically challenging to define and quantify the relationship between the ecosystem 'input', and a change in its quality and/or quantity, and the market good. Arguably, all market products and services are derived in some way from ecosystem inputs (e.g., water), but defining these relationships is complex, and the method is most suitably applied in cases where the ecosystem attribute / input is closely related to the market product in the production chain and contributes significantly to the output.

2.3.2.2 Non-market-based methods

Many of the benefits of ecosystems are not traded directly in a market. Several techniques have been developed for assigning economic values to non-market ecosystem benefits. One approach is to infer individual preferences for ecosystem benefits from observed consumer behaviour, for example, from the 'premium' people are willing to pay for a market good because of a particular set of ecosystem characteristics, such as a house with a pleasing view, or a tourism experience involving the presence of a particular ecosystem or species (Boyle, 2003; Whitehead et al., 2008). Techniques following this approach, termed revealed preference methods, are founded on a relationship of weak complementarity between ecosystem characteristics and the market good (Heal et al., 2005; Bateman et al., 2011) and are typically applied to ecosystem benefits related to recreation activities (using the travel cost method) and land / property values (using the hedonic pricing method) (Boyle, 2003). The defensive behaviour method (sometimes categorized as a cost-based approach) is conceptually different in that it is based on a substitute relationship, rather than a complementary one; ecosystem benefit values are derived from the actions (expenditure) consumers take to avoid or mitigate an environmental dis-amenity (Boyle, 2003) and include, for example, actions to reduce flooding or insurance taken to mitigate flood damage costs.

Underlying this approach is the assumption that a rational consumer will take action to avoid or mitigate damages as long as the value of the damages avoided is greater than the cost of the defensive action; these actions reveal the 'value' of ecosystem functions that limit or prevent such damages (Dickie, 2003).

Revealed preference methods apply statistical analysis to data on consumer's choices, generally drawing from a Marshallian demand function, and their application is limited to use values associated with ecosystems (Boyle, 2003; Whitehead et al., 2008). In South Africa, the travel cost method has been used to examine the relationship between river water quality and tourism value in the Kruger National Park (Turpie and Joubert, 2001) and value the recreational benefits of avoiding river inflow reductions to estuaries (Chege, 2009; Hosking, 2011). The hedonic pricing approach has been applied to determine the effects of the condition of an estuary on the value of near-by properties (Turpie et al., 2009). Expenditure on flood avoidance has been used as a proxy in valuing wetland and river rehabilitation (van Zyl et al., 2004). Revealed preference methods have not been frequently applied in the valuation of wetland restoration (Chapter 3). Published applications include the valuation of aesthetic benefits associated with wetland restoration (Earnhart, 2001, hedonic pricing) and recreational benefits associated with wetland creation (Alfranca et al., 2011, travel cost method).

In the South African context, the aesthetic and recreational benefits associated with wetlands vary with the context or setting, and, in some cases, wetlands are considered as a dis-amenity and their presence can negatively affect property prices, increase insurance premiums and reduce recreational use of an area. These dis-services are linked to the association of vegetation with hiding places for criminals, and wetlands as places for snakes and pests (e.g., mosquitos). On the other hand, wetlands can have significant recreational value associated with high biodiversity (especially birds) and hold cultural value related to ceremonies and sacred places, and, further, may be desirable for use value related to reed harvesting, livestock fodder and water and subsistence agriculture.

Stated preference techniques are another form of non-market valuation methods. The techniques are based on data – in the form of monetary amounts or other indicators of preference such as ratings or choices - from purposefully designed survey questions regarding the choices of individuals (or groups) when confronted with alternative states / scenarios of ecosystem condition and associated benefits (Brown, 2003). These surveys require people to directly state their preferences for ecosystem / environmental qualities (contingent approaches) or to choose between different bundles of ecosystem attributes designed to reveal their preferences (choice experiments) (Boxall et al., 1996; Whitehead et al., 2008). Stated preference approaches are the only methods currently available for assigning economic value to non-market (public) benefits. A further strength, in theory, of stated preference methods is their applicability to a range of ecosystem attributes and economic value types (use and non-use values) (Heal et al., 2005) and flexibility to explore various policy and / or ecosystem condition scenarios (Whitehead et al., 2008). Atkinson et al. (2012:30) argued, however, that in practice, stated preference methods encounter problems where people have little experience with, or understanding of, the object of valuation suggesting that: “Paradoxically, then, where SP techniques are most useful is also where they have the potential to be less effective”.

From an economic theory point of view, the hypothetical (in contrast to observed behaviour) nature of the approach is considered a primary weakness. The techniques assume that individuals understand the benefit in question and its relationship to ecosystem condition and will reveal their preferences (through choices or statements) in the hypothetical market just as they would in a real market. As stated by Whitehead et al. (2008:876); “At best, respondents give truthful answers that are limited by their unfamiliarity. At worst, respondents give trivial answers due to the hypothetical nature of the scenario”. Barbier (2007) warned of the difficulty in describing how changes in ecosystem structure and process affect the benefits or attributes valued by individuals.

Despite the controversy associated with stated preference methods, and their relative expense to apply, the review of the economic valuation of wetland restorations (Chapter 3) found stated preference methods to be the most common valuation approach applied. Westerberg et al. (2010) used a choice experiment to elicit preferences regarding land-use alternatives including wetland restoration. Pattison et al. (2011) and Lantz et al. (2013) both employed stated preference techniques to ‘value’ wetland retention and restoration alternatives. In South Africa, choice experiments have been used to examine the relationship between tourism and changes in river quality (Turpie and Joubert, 2001) and contingent valuation has been applied in assessing the preferences of recreationists regarding reduced flows (or averting reduced flows) to estuarine systems (Hosking, 2011). Challenges associated with applying stated preference methods in a rural, developing country context have been noted (e.g., Christie et al., 2012) and the importance of survey design and implementation emphasised in this regard.

2.3.2.3 Cost-based methods

Cost-based methods draw on various costs as a proxy for the value of ecosystem benefits and include methods based on the costs of damages incurred due to lost ecosystem functions / services (e.g., flood damage) or the costs of achieving the same benefit through an alternative means (replacement cost; e.g., buying feed for livestock as a replacement for wetland grazing in dry periods). Strictly, cost-based methods do not provide a measure of economic value as the WTP or WTA is not assessed. The approach assumes that if people incur damages (costs) as a result of a degraded ecosystem or pay to replace an ecosystem benefit then those ecosystem functions and benefits must be worth at least the costs that were incurred (Heal et al., 2005).

To improve the validity of replacement cost estimates as a proxy of the value of ecosystem benefits, several application conditions have been defined: (i) the replacement must be equivalent in quality and magnitude to the ecosystem service (perfect substitute); (ii) the replacement must be the least cost option of replacing the service; and (iii) people must be willing to pay the replacement cost to obtain the service (i.e., there must be a demand for the services / benefit) (Shabman and Batie, 1978). Where the conditions do not hold, cost-based approaches are unlikely to accurately reflect the economic value of the ecosystem benefit. Cost-based methods are most appropriately applied where damage costs or replacement expenditures have been incurred, in which case the cost-based estimate reflects a lower-bound measure of the value of the ecosystem benefit (Brouwer et al., 2013).

The application of cost-based approaches is often regarded as easier and cheaper to estimate, and, arguably, simpler to explain than other economic valuation methods (Dickie, 2003). This is reflected in their relatively more frequent application in wetland intervention valuation assessments compared to alternative methods, except for stated preference approaches (Chapter 3). A common application of the replacement cost approach is to use the cost of built infrastructure-based water treatment as a proxy of the value of water filtration benefits associated with wetlands (Pagiola et al., 2004). 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 (USA) 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. Barbier (2007) suggested that it was sufficient for the policy decision simply to demonstrate the cost-effectiveness of the restoration and protection option compared to the alternative.

In South Africa, the replacement cost approach has been applied in estimating the value of wetlands in ameliorating water pollution in the Western Cape (Turpie et al., 2010). Van Zyl et al. (2004) valued the flood attenuation benefits of urban river and wetland rehabilitation using flood damage costs as a proxy. Cost-based methods are also used in the field of health economics, where a 'cost of illness' approach is used as an indication of the value of ecosystem functions that prevent or limit illness (Barbier, 2007). In South Africa, de Lange et al. (2012) employed the 'cost of illness' method to evaluate the impacts of microbial water pollution.

2.3.2.4 Value transfer

The value transfer approach is not a specific economic valuation method, but rather a term adopted to refer to the use of information from available valuation studies in a different context, in other words, a form of secondary valuation. The 'secondary' valuation involves the 'transfer' of value information from primary valuation studies of similar systems or sites to the site under consideration (Rosenberger and Loomis, 2003; Brouwer et al., 2013). For example, 'transferring' the recreational value of bird viewing estimated for a specific wetland to reflect or indicate the (potential) value the same service of another or across multiple wetlands. Valuation information may be transferred directly as per unit value estimates (value-estimate transfer); as value estimates adjusted for the new site / context; or as the functional form or statistical model (value-function) estimated in the primary valuation study (Heal et al., 2005; Turner et al., 2008).

Value-estimate and value-function transfers are only as robust as the original studies on which they are based and the correspondence in context between the primary and secondary sites. There are several sources of error associated with the application of the transfer approach: (i) measurement error, which is related to bias and assumptions in the primary study (or studies, in the case of meta-analysis); (ii) generalisation error, which emerges from disparities between the primary study and the new site; and (iii) publication selection bias, that is, error as a result of bias in the empirical literature base linked to publication selection criteria (e.g., statistically significant results, application of 'new' methods) (Rosenberger and Stanley, 2006). The value-function transfer approach is considered more accurate than value-estimate transfer as it allows some of the characteristics of the functional relationship to be adjusted to better suit the 'new' site whereas value-estimate transfer is invariant to differences between the original and the 'new' site (Rosenberger and Loomis, 2003).

Confidence in the estimates generated using the value transfer approach depends on the availability and quality of primary valuation studies and the degree of similarity between the original site and the new / policy site. Confidence is improved when the circumstances in the new context are similar to the prevailing characteristics of the original research, for example, when the ecosystem condition, location, ecological change (and its implications) and affected population are similar for the two sites and / or when the transferred value can be adjusted to reflect important differences between sites; and when the primary valuation study was rigorous and applied appropriate valuation techniques (Vieira et al., 2014). The transfer method is generally applied when time and resources are not available for primary site-specific valuation. Heal et al. (2005) noted the frequent use of the transfer approach in policy analyses. However, given the complex and context specific nature of ecosystem function and benefit relationships, the value transfer approach is characterised by low confidence (high uncertainty) (Brouwer et al., 2013). A comprehensive and wide-ranging set of primary empirical studies is required to support the value transfer approach (Rosenberger and Stanley, 2006).

Value transfer is regularly applied in the valuation of wetland restoration, with value-estimate transfer more common than value-function transfer (Chapter 3). Vieira et al. (2014), for example, applied value transfer, drawing from both a meta-analysis study of wetland values and a primary study, to value the benefits of coastal wetland restoration. Adusumilli (2015) applied a value-function approach, developed from a meta-analysis of wetland ecosystem service values, in the valuation of ecosystem services from wetland mitigation projects. While much debate remains on the reliability of value transfer, the approach is a practical and relatively affordable means of generating economic value information for wetland restoration. The reliability and scope of application of the transfer approach can be increased by establishing a base of robust primary valuation studies across a range of locations and contexts. However, such an empirical (published) evidence base of wetland restoration valuation studies is lacking in South Africa (Chapter 3).

2.3.2.5 Economic valuation methods for wetland benefits

Certain valuation methods are applicable only to particular benefit or value dimension (e.g., market vs. non-market benefits, use vs. non-use values), while other methods can be applied more broadly. Several different methods may be applied in a single assessment to cover a range of benefit types, values and affected groups (beneficiaries). However, the application of multiple methods can give rise to 'double-counting' benefits and overestimating the values (Boyle, 2003). On the other hand, it is often the case that only a sub-set of the restoration benefits are 'valued' in a single study (Chapter 3).

In Table 2.3 the wetland services identified in the WET-EcoServices tool (section 2.2.2.2) are linked with relatively more suitable, or commonly applied, standard economic ecosystem valuation methods. Potential benefit outcomes, economic value indicators and data requirements are also identified; these are presented as examples, rather than as an exhaustive list.

Table 2.3: Wetland ecosystem services, benefit outcomes, economic value indicators and economic valuation methods

Wetland service	**Benefit outcomes**	**Economic value indicator examples**	**Economic valuation methods and example data requirements**
Economic valuation methods: AD - avoided costs; H – hedonic pricing; M – market based; RC - replacement cost; SP – stated preference; TC – travel cost.			
Flood attenuation	Reduction in downstream flood danger (risk). Improved/sustained flood protection for downstream infrastructure (e.g., roads, fences, houses) and land use (e.g., agriculture, livelihood / subsistence activities, aesthetic and recreation).	Difference in water levels / flood damage with and without rehabilitation. Trends in flooding events (e.g., number and severity of floods over time). Number of people residing in 'flood zone'. Extent of infrastructure and land-use downstream at risk of flood damage with and without rehabilitation.	Methods: AC, RC, H Reduction in risk and/or area protected as a result of rehabilitation. • Change in flood lines • Flood discharges & associated floodwater levels, flow data, probabilities of flooding. Downstream infrastructure and land-uses. Value of damages to downstream infrastructure, property and livelihood activities in the case of no rehabilitation. • Previous flood damages, frequency and costs. Difference in property values with and without rehabilitation due to reduced risk of flood damage and/or property improvements. • Property prices over time and expert inputs from property valuers. Difference in insurance premiums associated with flood risk with and without rehabilitation. Cost of mitigation measures with and without rehabilitation; costs of alternative ways (e.g., built infrastructure) to achieve the reduced level of flooding with rehabilitation compared to without rehabilitation.

Wetland service	Benefit outcomes	Economic value indicator examples	Economic valuation methods and example data requirements
Economic valuation methods: AD - avoided costs; H – hedonic pricing; M – market based; RC - replacement cost; SP – stated preference; TC – travel cost.			
Streamflow regulation	Increased streamflow during low flow periods - extended period of baseflow. Improved downstream water supply during dry season/periods (for wetlands linked to watercourses). Largely supports enhanced provision of other services. Water available during low flow/rainfall periods (that would otherwise not be available) for domestic, agriculture or other purposes, in situ & downstream (e.g., fishery production). Improved / extended habitat and biodiversity maintenance and soil moisture (into dry season).	Difference in base flows (volume and / or timing) with and without rehabilitation. Downstream water use and numbers of users and dependence on the resource (i.e., availability of alternatives). Use / demand for other linked services (e.g., crop cultivation into the dry season).	Methods: M (production function), AD, RC, SP, H Information on onsite (surrounding) and offsite (downstream) water use and land-use activities, effect on productivity and production data (e.g., crop or fisheries production). Alternative sources of water during low flow periods and associated costs to access (e.g., boreholes). Alternative water storage options (to store the volume difference) and associated costs. Costs of built infrastructure (e.g., water tanks) to store the volume difference of low flows with and without rehabilitation. Beneficiary preferences regarding outcomes. Effect on downstream property values associated with increased / reduced stream flows.

Wetland service	Benefit outcomes	Economic value indicator examples	Economic valuation methods and example data requirements
Economic valuation methods: AD - avoided costs; H – hedonic pricing; M – market based; RC - replacement cost; SP – stated preference; TC – travel cost.			
Water quality enhancement	Improved water quality for downstream uses. Bulk water supply Improved downstream recreation cultural opportunities and aesthetic benefits. Human health benefits / reduced health risk (e.g., recreation and irrigation). Reduced animal health impacts (or risk) and associated economic and livelihood impacts. Reduced damages associated with use of poor quality water (e.g., damage to pipes, soil fertility impacts). Reduced sedimentation of downstream streams and dams.	Difference in quantity of pollutant in downstream flows with and without rehabilitation. Types of water use, numbers of users, dependence on the water resource. Wetland abundance in the same catchment.	Methods: RC, AC, SP, H Alternative (realistic) water treatment options and associated costs to achieve the same level of water quality enhancement. Access to and costs of alternative water sources/supplies. Damages and associated costs from using poor water quality (human and animal health impacts, dam storage capacity, impacts on irrigation equipment, impacts on production). Mitigation measures and costs to avoid damage from poor quality water (e.g., sediment filters in irrigation systems, dredging of dams). Beneficiary preferences regarding outcomes. Potential property value effects - difference in values of residences located near the wetland, and downstream, with and without rehabilitation due to improved water quality (aesthetics and odour). • Property prices over time and expert inputs from property valuers.

Wetland service	Benefit outcomes	Economic value indicator examples	Economic valuation methods and example data requirements
Economic valuation methods: AD - avoided costs; H – hedonic pricing; M – market based; RC - replacement cost; SP – stated preference; TC – travel cost.			
Carbon storage	Increased retention of carbon. Increased contribution to global climate regulation. Mitigation of carbon released through wetland degradation.	Difference in 'total carbon' stored in the wetland with and without rehabilitation. The above difference in total carbon converted into Equivalent Total CO_2.	Methods: M, SP Total amount of carbon stored in the wetland with and without rehabilitation / difference in Equivalent Total CO_2 • Soil analysis data • Significance in terms of global carbon emissions. Carbon trade prices; global social cost of carbon (Nordhaus, 2017) and South Africa's share of this cost based on proportional GDP contribution and vulnerability index (e.g., Turpie et al., 2017). Beneficiary preferences regarding outcomes.
Provisioning services Provision of water for human use, harvestable resources, livestock fodder, cultivated foods	Increased amount of the good, or length of season the good is available, for domestic, agricultural or other purposes. Health benefits (human and animal – medicinal plants used in animal husbandry; water for sanitation; food diversity). Enhanced/sustained income generating activities. Cash expenditure avoided.	Difference in the amount of the good available, or timing of availability, with and without rehabilitation. Number of users/beneficiaries. Availability and/or access to alternatives or substitutes; reliance directly on the wetland as the source of the good Improved livelihood opportunities (e.g., income from sale of additional quantities of foods). Provision of water is closely related to streamflow regulation service benefits (potential source of double-counting).	Methods: M, RC, SP Difference in the amount of the good available, or timing of availability, with and without rehabilitation. Quantified input-output relationships in the case of the wetland good as an input to an economic good. Access to alternatives and costs associated with securing alternative sources (e.g., time and costs to collect water from another source). Socio-economic information of users/beneficiaries. Beneficiary preferences regarding outcomes.

Wetland service	Benefit outcomes	Economic value indicator examples	Economic valuation methods and example data requirements
Economic valuation methods: AD - avoided costs; H – hedonic pricing; M – market based; RC - replacement cost; SP – stated preference; TC – travel cost.			
Cultural services Cultural and spiritual experience Tourism and recreation Education and research	Enhanced opportunities for cultural and spiritual experiences, tourism and recreation, and education and research • Including enhanced aesthetic outcomes and enhanced protection of cultural heritage. Increased economic opportunities associated with tourism and recreation. Improved practice, awareness raising, skills and eco-literacy development as a result of enhanced education and research activities.	Differences in attributes influencing cultural and spiritual experiences (including aesthetic appreciation), tourism and recreation opportunities and education and research. Numbers of users/beneficiaries. Surrounding land-uses – communities residing nearby and their cultural orientations, dependence on the wetland for fulfilment of 'cultural' needs/wants. Availability and/or access to alternatives or substitutes. Designation as a specifically recognised site of importance (e.g., as a nature reserve, world heritage site, RAMSAR site).	Methods: SP, TC, H, RC Identification of differences with and without rehabilitation in attributes relevant to cultural benefits Beneficiary preferences regarding outcomes - people's preferences and willingness to pay for, or accept compensation for, cultural benefits (associated with both use and non-use values) Differences in number of tourists and their expenditure when visiting the wetland with and without rehabilitation. • Tourism statistics, visitors spending and travel costs. Availability and/or access to alternatives or substitutes and associated costs. • Wetland abundance in the region and (cultural) uniqueness of the wetland in comparison with other nearby wetlands • Entrance fees to similar wetlands Value of spending on research and / or educational activities. Potential property value effects – difference in value of properties located near the wetland with and without rehabilitation related to aesthetic benefits for residents or accommodation opportunities associated with improved tourism and recreation opportunities. • Property prices over time and expert inputs from property valuers.

2.3.3 Challenges to theory and practice

Views on the use and appropriateness of economic ecosystem valuation are diverse. Concerns stem from ideological and technical challenges associated with the assumptions and limitations of neoclassical economic theory, practical difficulties in application, and questions concerning the social goals that guide decision-making. Several of these challenges and perspectives are discussed in the following sections.

2.3.3.1 Assumptions and limitations of neoclassical economic theory

There are many critics of the neoclassical economic valuation approach (e.g., Kant, 2003; Chee, 2004; Nelson, 2008; Sagoff, 2008; Foster and Clark, 2009; Parks and Gowdy, 2013; Spash and Aslaksen, 2015) and the dominance of the economic metaphor of the human-nature relationship is widely contested (Norgaard, 1989; Martinez-Alier, 2008; O'Neill et al., 2008; Luck et al., 2012; Jax at el., 2013; Fanny et al., 2014; Diaz et al., 2015; Aries-Arevola et al., 2018).

Neoclassical economics is underpinned by a particular ontology and epistemology leading to a narrowly defined theory of value and a specific conception of human behaviour. There is an extensive literature arguing that the neoclassical economic model is limited as an explanation of human behaviour and too restrictive in its conceptualisation of value including as a way of articulating environmental values (section 2.1). Several points are noteworthy in reflecting on the assumptions and limitations of ecosystem economic valuation.

- Valuations are not ideologically neutral, they reflect specific ontologies, institutional arrangements and historical and existing power structures, all of which influence the way we think about the human-nature relationship and what and how we 'value' (Martinez-Alier, 2002; Røpke, 2005).
- Ecosystem values are multiple and not commensurate (Vatn and Bromley, 1994; Martinez-Alier et al., 1998) and the neoclassical economic valuation framework is incapable of capturing the manifold of value dimensions associated with nature (Chan et al., 2012a; Spangenberg and Settele, 2016), as is any single framing or approach[32].

[32] Ecosystem 'value' is a complex notion associated with a multitude of meanings, theoretical perspectives and ideological dimensions; value incommensurability and value pluralism are increasingly recognized in the field of environmental valuation.

Economic valuation methods applied alone limit the consideration of ecosystem values to that of economic value, yet many ecosystem structures and processes are not amenable to economic valuation[33] and / or hold other types of values (Blignaut et al., 2007; Rees et al., 2007; Sagoff, 2008; Turner et al., 2008).

- Economic ecosystem valuation is based on marginal utility which assumes small, non-critical, changes in the underlying stock, however, ecosystems are characterized by ecological thresholds and regime changes (Crépin et al., 2012; Pelenc and Ballet, 2015) which are, along with the consequences when they are crossed, difficult to predict (Blignaut et al., 2007; Farley, 2012). Marginal value estimates become unreliable in the case of critical changes in the ecosystem as a result of reaching / crossing a threshold level.
- Neoclassical economics is underpinned by the rational actor model of human behaviour and its associated assumptions. Accumulating evidence from across a range of fields, however, challenges this model as a realistic representation of human behaviour (e.g., Henrich et al., 2001). Many of the critiques of neoclassical economics are not new (e.g., Pearce, 1976; Georgescu-Roegen, 1975; Sen, 1977; Westman, 1977), but remain unresolved (Baveye et al., 2013).
- Research, for example, from behavioural economics, psychology and neuroscience, provides evidence of human behaviour motivators beyond individual preference satisfaction, such as fairness, reciprocity, moral duty, and community or cultural norms[34] (Howarth, 1995; Henrich et al., 2001; Gowdy, 2004).
- The rational actor model assumes that individuals have perfect knowledge about outcomes on which to judge consequences. Many ecosystem processes are challenging to discern and few people, if any, are fully aware of the consequences of ecosystem change for humans (Naess, 2006a). Important outcomes may be overlooked out of ignorance[35].

[33] Sagoff (2008), for example, highlighted that some ecosystem services are 'lumpy goods' that cannot be valued incrementally (i.e., goods / services that cannot be divided in parts, or sold in units).
[34] Spash (1997) for example, put forward evidence of a rights-based (deontological) motivation, rather than a purely utilitarianism-based approach, to decision-making.
[35] Further, there are limits to individual information processing in terms of information availability, individual cognitive capabilities and time (Kahneman, 2003; Samson, 2016).

- There are fundamental moral judgements inherent in economic valuation (Røpke, 2005), and ethical concerns beyond instrumental human interests (O'Neill and Spash, 2000; Costanza, 2000; Liu et al., 2010). In conventional CBA, ethical judgments underpin the rationale of discounting.
- The neoclassical approach to measuring social welfare is the aggregation of individual welfare (based on individual preference satisfaction), yet the objectives and values of a society can differ from those of its individuals (Swaney, 1987; Kenter et al., 2015)[36].
- The notion of substitutability is inherent in the concept of economic exchange value and the neoclassical economic model takes a weak sustainability position (Blignaut et al., 2007; Luck et al., 2012 Spangenberg and Settele, 2016) and is optimistic about the ability of technology to compensate for the degradation of ecosystems (Rees et al., 2007).
- Ecosystem and land-use decision situations are complex, often characterised by conflict, and cannot be reduced to single value frameworks (Røpke, 2005; O'Neill et al., 2008).
- Ecosystems have emergent properties – properties of the system that arise from interactions between the 'smaller' parts which are not present when the 'parts' act alone. Economic valuation is seldom adequate to reflect the importance of these emergent properties and broad-scale interactions (Turner et al., 2008; Gómez-Baggethun, 2010). Assigning value to the 'services' of wetlands is not equivalent to articulating the importance of intact wetlands themselves, and their interactions with other elements within a broader ecological system.

Nelson (2008) suggested that few practising economists consider typical human behaviour to fall solely within the rational actor model. This is evident in the emergence of a number of expanded or alternative schools of economic thought such as institutional and behavioural economics. In the opinion of Parks and Gowdy (2013:e6) "[c]urrent theory and empirical research in economic valuation has moved well beyond the basic expected utility models but these ideas have been relatively slow to influence environmental valuation techniques and

[36] Feldman (1987:894) argued that "there is no logically infallible way to aggregate the preferences of diverse individuals" (quoted by Parks and Gowdy, 2013).

policy analysis”. Despite the controversies associated with neoclassical economics, the economic valuation of ecosystem services is “increasingly seen as a crucial element of robust decision making”, reflected in a growing body of related research (Atkinson et al., 2016:22). On the other hand, Laurans et al. (2013) found little evidence that such valuations guide specific action-taking, but rather that ecosystem service values are used rhetorically for general influence and raising awareness.

2.3.3.2 Divided perspectives

Standard economic valuation is poorly equipped to address many of the characteristics of ecosystem change evaluations, including strong uncertainty and knowledge limitations, irreversibility and non-marginal changes, limited substitutability, and the diversity of human behaviour motivators and value dimensions. For some, these limitations mean that the scope of economic ecosystem valuation is narrow; it is appropriate only in certain situations or under specific conditions (e.g., Bockstael et al., 2000; Kallis et al. 2013; Fanny et al., 2015) and to be used as a complement to other forms of valuation and ethical and scientific arguments for conserving or restoring ecosystems (Norton and Noonan, 2007; Martín-López et al., 2014). Others eschew its use at all. Sagoff (2008:252), for example, declared: “By 'putting a price on it' we abandon the rhetoric of reverence; we regard nature as a resource to exploit rather than a heritage and an endowment to maintain”. Several thinkers call for a meaningful transformation of the standard economic approach to support ecologically considerate decision-making (e.g., Norgaard, 1989; Costanza, 1989; Holland, 2002; Wight, 2012; Spash and Aslaksen, 2015; Costanza et al., 2017).

The limitations of neoclassical economic theory are recognized within the economic discipline, and dissatisfaction with the orthodox approach has led to various schools of thought that expand on, or diverge from, neoclassical economics. New institutional economics moves away from the neoclassical assumptions of perfect information and bounded rationally, recognizing, as a result, that decisions and actions are subject to uncertainty (Menard and Shirley, 2008). The study of institutions and their interactions with organizational arrangements is the basis of new institutional economics. This focus has led to a particular interest in behavioural models and the ways humans interpret reality with attention to how these ontologies shape the institutional environmental (Menard and Shirley, 2008). New institutional economics is concerned with explaining change (economic, political and social) and understanding the underlying beliefs, norms and incentives that govern human interactions and drive change. Interactions, the institutional and cultural framing of decisions and actions, information, cognitive limits and transaction costs, and the influence of context and method on preference and value formation are key concepts of new institutional economics (van Kerkhoff and Berry, 2016). It is for these areas of emphasis that Menard (2011:119) proposed that new institutional economics can "provide powerful tools and useful insights in analyzing environmental problems and assessing potential answers".

Feminist economics perspectives question the growth-based capitalist paradigm of neoclassical economics (Perkins, 2007). Feminist economics is closely tied to issues of justice and equality and concerned with what is broadly termed as 'social provisioning'[37] (Power, 2004). Feminist perspectives and research have been put forward as offering insights and frameworks that bridge social and ecological issues, particularly insights on non-monetized exchange relationships, justice, provisioning, valuation, diversity, citizenship, and wealth and power concentration, among others (Perkins, 2007, Nelson, 2008; Spencer et al., 2018).

Behavioural economics challenges the rational actor model of neoclassical economics, particularly the assumptions of unbounded rationality and self-interest (Samson, 2014). Human decision-making is the primary subject matter of behavioural economics, which emphasises that people do not always act rationally; their decision-making is affected by context, restricted by information limits and cognitive capacity and influenced by social norms (Kahneman, 2003; Samson, 2014). Alternative perspectives on human behaviour and insights from behavioural economics are seen as significant in the development of ecologically considerate policy (van den Bergh et al., 2000; Venkatachalam, 2008; Carlsson and Johansson-Stenman, 2012).

[37] Social provisioning is particularly concerned with caring and unpaid labour as fundamental economic activities, well-being as a measure of economic success, economic, political, and social processes and power relations, ethical goals and values and the interrogation of differences by class, race-ethnicity, and other factors (Power, 2004).

Cultural economics also has insights to offer environmental decision-making, both in challenging neoclassical perspectives of value and valuation and in conceptualising 'intangible cultural capital' and addressing the complex ways in which individuals and societies value cultural goods (Throsby, 1999, Klamer, 2002; 2003). Cultural 'services' are included in all major ecosystem service typologies, but are particularly challenging to characterize and value from a neoclassical economics perspective (Chan et al., 2012a). Cultural and environmental economics deal with similar issues, particularly in terms of valuing intangible cultural benefits and non-use values.

Ecological economics takes the view that the economic system is a part of, in contrast to being distinct from, the ecological system (Costanza et al., 1997a); 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 of ecological economics (Costanza et al., 1997b). Ecological economics is intended as an approach that integrates the thinking and methods of many disciplines in addressing the goals of sustainability, fairness and efficiency (Costanza et al., 1997a). Both value and methodological pluralism are advocated (Norgaard, 1989; Venkatachalam, 2007; Gómez-Baggethun and Barton, 2013). Conventional neoclassical economics is regarded as one input to a broader transdisciplinary synthesis (Costanza et al., 1997a). In reflecting on ecological economics over the previous 20 years, Costanza et al. (2017:1) concluded that much of the research and debate highlights "the weakness of the mainstream economic approaches to valuation, growth, and development". Given that valuation is inevitable in decision-making, Costanza et al. (2017:7) ask then "what kind of valuation is most appropriate?".

Clive Spash and colleagues suggested that the economic valuation of ecosystems is proceeding in three 'camps' (Spash, 2013; Spash and Aslaksen, 2015). One camp is that of the 'New Environmental Pragmatists'[38] championed largely by environmentalists, ecologists and

[38] The other two 'camps' are that of the New Resource Economists who view 'ecological economics' as grounded in neoclassical economics, and the Social Ecological Economists who call for a transformation of neoclassical economics and aim to address its fundamental weaknesses.

conservation biologists[39], who, argued Spash and Aslaksen (2015), are not familiar with the foundations and assumptions of economic theory[40]. The aim of 'new environmental pragmatists' is to raise the policy profile of ecosystem degradation and motivate for ecological restoration and conservation. To this end, economic valuation is viewed as a way to generate "communicatively powerful statements of why everyone should be paying more attention to environmental problems" (Spash, 2013:355). Broadly, the approach of this 'camp' is to adopt "methods and concepts because they are deemed to be effective under current political conditions and economic institutions" (Spash, 2013: 354). However, Spash (2013) and Spash and Aslaksen (2015) contend that this pragmatic approach gives little attention to the theory underpinning the economic framework and the limitations of the standard economic model, leading to an "over extension of environmental economics" and the employment of "ever cruder methods" (Spash and Aslaksen, 2015:248), to the extent that 'New Environmental Pragmatists' "have employed cost–benefit tools in ways practicing environmental economists would never have dared to do" (Spash, 2013:360).

In a review of trends in the development of ecological economics, Ropke (2005) identified a similar divergence in views within the field, noting that "some (mainly economists) see ecological economics as a contribution towards changing the discipline of economics in a radical way", whereas "others (mainly natural scientists) look for more immediate influence on the political agenda, concentrating on core messages regarding the seriousness of the environmental situation and trying to come up with illustrative numbers" (Ropke, 2005:281). Spash and Aslaksen (2015:252) are somewhat more scathing in their description, stating that "[e]conomic theory has limitations, and supposed pragmatism which ignores them can only produce meaningless numbers for rhetorical purposes". The authors conceded, however, that many 'new environmental pragmatists' recognise that the economic value estimates they generate may lack theoretical rigour, but that this is justified as the 'numbers' draw attention in the decision-making arena. Rees et al. (2007), for example, noted that even though economic value estimates may be imprecise and underpinned by controversial methods, an

[39] The quantitative literature review of wetland restoration valuation undertaken as part of this research (Chapter 3) indicated that in 43% of the publications none of the authors appeared to be affiliated with an economics related department or institution.
[40] Whether all practicing economists have this 'familiarity' is also questionable.

inventory of benefits reported with assigned economic values, draws attention to the range of positive outcomes of ecosystem maintenance / restoration and can be influential in motivating restoration interventions. Gómez-Baggethun et al. (2010) found that economic ecosystem valuation has contributed to attract political support for conservation. Parks and Gowdy (2013) warned, however, of the risk of poorly designed valuation studies and theoretically weak value estimates in discrediting the effort to articulate the importance of ecosystem structure and function to human well-being and the economy.

Clive Spash and colleagues (e.g., Spash, 1993; 1997; Spash and Ryan, 2012; Spash, 2013; Spash and Aslaksen, 2015) challenged the rationale of economic ecosystem valuation and standard economic concepts for communicating the importance of nature and as a 'solution' to environmental degradation and suggested that there is "much conjecture in this position and a lack of reflection upon the literature covering human motivation, environmental values and ethics, political science and institutions" (Spash and Aslaksen, 2015:251). Jacobs et al. (2016:215) suggested that the complexity of valuation and the challenges of practical application "defy hopes for a methodological silver bullet" and concluded that a combination of perspectives, disciplines and methods is needed to reflect the diverse values of nature, noting however, the difficulty in communicating "value complexity and uncertainty in a comprehensive and compact way that can be easily digested for use by practitioners and decision-makers".

2.3.3.3 Unintended consequences

Given its pervasiveness and influence, the economic framing has the potential to significantly shape humanity's perception of value and the human-nature relationship (Maki, 2011; Spash and Aslaksen, 2015). It is this 'influence' that many environmentalists and ecologists wish to harness in motivating for ecosystem preservation and restoration. However, reliance on the economic value framing of nature has latent consequences. While economic ecosystem valuation has been intended to illuminate the importance of intact ecological systems, to reduce degradation and justify restoration; the economic framing also influences the way people perceive and relate to the non-human world (Gómez-Baggethun et al. 2010). As argued by Maki (2012:xvi) "…in society at large, strong 'economistic' trends (of marketisation, commercialisation, commodification, monetisation) increasingly shape our cultural and mental landscape, and the discipline of economics relates to these processes both as a spectator and as a contributor".

With the introduction of a specific valuation approach, a particular framing and rationality is imposed on the formation and development of values (Gómez-Baggethun and Ruiz-Pérez, 2011). Several theorists warn that favouring an economic valuation approach can obscure other types of value and exclude non-economic motivations for protecting ecosystems[41] (Claro, 2007; Chan et al., 2012a; Luck et al., 2012; Jax et al., 2013). Frey and Jegen (2001:590) have argued that management practices based solely on the economic model "may dampen (or even destroy) non-economic realities such as intrinsic motivation and social relations".

Once the idea of the 'economic value' of ecological restoration has been introduced as a justification, it may be difficult to 'elicit' other value types or articulate other forms of human-nature interactions[42]. Klamer (2003) warned of the attraction of formal algorithms and single value metrics – such as economic value – to decision-makers who perhaps seek a way to distance themselves from taking 'difficult' decisions by applying technical calculations, risking the displacement of other values, goals and forms of knowledge. A risk being, that if the economic valuation of ecosystems and ecological restoration dominates other dimensions of value and societal goals, ecosystem structure and function lacking economic (exchange) value, or where the challenges of estimating economic value are prohibitive, may be ignored in decision-making, arguably perpetuating the 'problem' economic ecosystem valuation attempts to address[43].

Gómez-Baggethun and Ruiz-Pérez (2011) view economic valuation as a driver of commodification while Spash and Aslaksen (2015) refer to the 'neoliberalisation of nature'. These ideas resonate with the tension between ecological limits and continued economic growth and capital accumulation (Arsel and Büscher, 2012). On the one hand, is the discourse of sustainable economic growth, the idea that long-term economic growth can be 'de-

[41] Many of these warnings draw from Motivation Crowding Theory (see Frey and Jegen, 2001).
[42] However, as noted by Vatn (2010) selecting or prioritising evaluation processes is a conundrum - 'the second order problem' – which requires further attention in environmental appraisal.
[43] That is, the 'problem' that the contributions of ecosystems, and the consequences of ecosystem degradation, remain 'unpriced' and left out of conventional economic cost-benefit analysis processes.

coupled' from negative environmental consequences, epitomised, for example, in the 'green economy' concept adopted by the South African Department of Environmental Affairs: "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). On the other hand, are arguments that continued economic growth is incompatible with ecological and equity concerns and calls for attention to reducing and redistributing consumption levels (Naess, 2006b).

Several theorists have argued that a consequence of a disproportionate focus on ecosystem valuation is a lack of attention to the institutional changes needed to address the drivers of ecosystem degradation (Norton and Noonan 2007; Norgaard, 2010; Spash, 2013; Spash and Aslaksen, 2015). Clive Spash and colleagues view the drivers of ecosystem degradation and biodiversity loss as "structural problems within the dominant capital accumulating political economy" (Spash and Aslaksen, 2015:251) and have called for "a fundamental critique of the dominant structure of political economy and its treatment of human relationships with Nature" (Spash, 2013:355). Spash and Aslaksen (2015:252) proposed shifting the ecosystem valuation debate to focus on identifying "the best institutions humanity can create that are able to articulate different values, empower silent voices and the disenfranchised, and recognise and address issues of injustice and abuse of power"; institutions that enable human-nature relationships to "evolve in a sense of care and respect rather than exploitation and dominance".

2.3.3.4 Technical challenges

In addition to the limitations of, and objections to, neoclassical economic theory, the application of economic valuation to ecosystem processes and attributes has proven technically challenging (Heal et al., 2005; Nelson et al., 2009). By their nature, ecosystems pose two principal practical challenges for economic valuation: system complexity (Turner et al., 2000; Norgaard, 2010) and the atypical nature (as economic commodities) of the benefits derived from ecosystems (O'Neill and Spash, 2000; Fisher et al., 2008; Wenger and Pascual, 2011). Complexity relates to the dynamic, unpredictable, interconnected system characteristics of ecosystems and their properties of emergence, non-linearity, limited substitutability, irreversibility and threshold effects (Bockstael et al., 1995; Holling, 1996; Wallington et al., 2005; Schoon and van der Leeuw, 2015). Further, ecosystems and human systems are interconnected and interdependent, influencing and shaping one another. The view of human and non-human systems as inextricably linked is evident in socio-ecological systems (Anderies et al., 2004) and social-ecological relations (Basset and Piemer, 2015) thinking.

This complexity, and our incomplete knowledge of how ecosystems function, means that the relationships between ecosystem structure and process and the benefits derived by humanity are not simple to define and measure (Daily et al., 2000; Heal et al., 2005; Polasky and Sergerson, 2009; Barbier, 2013; Costanza et al., 2017). Biophysical processes (e.g., nitrate uptake by wetland vegetation) are seldom the attributes directly appreciated by people (e.g., water of a quality suitable for drinking) (Keeler et al., 2012). These biophysical processes need to be 'translated' or linked to attributes that people can relate to and value. A fundamental challenge for the economic valuation of wetland restoration is defining and quantifying the relationship between wetland structure and process and human well-being outcomes. The ecosystem services framework has been proposed as a way of conceptualizing the pathway between ecosystem structure and process and human well-being (section 2.2.2).

A second central challenge in the application of economic valuation to ecosystem-derived services is that of assigning economic value to benefits characterised by properties atypical of conventional economic commodities. In the case of ecosystem benefits, such properties are particularly related to the public good and often intangible nature and many non-use values of ecosystem benefits. Public goods are products and services that are non-excludable and non-rival[44]. Many of the supporting and regulating services of ecosystems are, to various degrees, public goods (Farley, 2008). Scarcity is a key element in defining an economic good; 'goods' that contribute to satisfying human wants for which the demand exceeds supply are economic goods and hold value in exchange. Goods that are unlimited in supply are regarded as non-economic and hold no direct exchange value (market price). Scarcity, and therefore,

[44] While Farley (2008) suggests climate stability and the ozone layer are examples of public goods, it could be argued that these are no longer strictly non-rival, given the negative outcomes of the use of the atmosphere as a 'waste sink'.

the concept of non-economic goods, is relative to situation and time and goods may be ‘non-economic’ in one context and 'economic’ in another. Economic value estimates are context specific. Further, ecosystems and biodiversity are regarded as having non-use value, yet there is much debate on how these values arise and whether they can be articulated and measured using conventional economic valuation methods (Crowards, 1997; Atkinson et al., 2012).

In the context of ecosystem restoration, economic valuation is further complicated by the need to isolate the specific influence of the restoration actions on ecosystem structure and process and how this translates to a difference in ecosystem service flows compared to the case without restoration. The economic valuation of wetland restoration is concerned with evaluating well-defined changes in the ecosystem that result from the restoration (Bockstael et al. 2000; Pagiola et al. 2004). This involves comparing the ‘with’ and ‘without’ restoration cases and, to the extent possible, isolating the effects of the restoration from other factors affecting the supply and demand of ecosystem services and the attendant benefits (Freeman, 2003; Pendleton and Baldera, 2010). The ecological response to the restoration must be clearly linked to a human well-being outcome (benefit), which can then be valued (Jenkins et al., 2010; DEFRA, 2011; Vieira et al., 2014)[45].

Predicting ecological response and defining ecosystem function – human well-being benefit relationships are not simple tasks. Defining the relationship between ecosystem structure and process and the flow of ecosystem services and how changes in ecosystem condition affect the flow of services for wetland systems is hampered by our lack of ability to predict, model and quantify these relationships (although this is increasing) (Barbier, 2011; Suding, 2011; Barbier, 2013). The very nature of ecosystems makes the task of predicting the outcomes of restoration with a reasonable degree of accuracy challenging (Daily et al., 1997; Bockstael et al., 2000). Added to the difficulty of this task is the specificity of wetland services, and the related benefits, to location and situation (Mitch and Gosselink, 2015), limiting our ability to isolate generic characteristics or develop general principles (Heal et al., 2005). As noted by Milon and Scrogin (2006:172): “Policy analysis for wetland ecosystems is especially difficult

[45] These main aspects or components of economic ecosystem valuation were identified through a review and analysis of ecosystem valuation frameworks and guidelines (see Appendix 2.1 for a table of references) and build on knowledge presented in earlier sections.

because these systems provide multiple, interdependent services that vary by type of wetland, location, ecohydrological management, and other factors".

Wetland services are not necessarily beneficial across all contexts or individuals, for example, "an increase in water quantity is a dispassionate description of a change in volume; this increase might be beneficial in the context of diverted water supply and detrimental in the context of flood damage" (Brauman et al., 2007:72). These nuances create challenges for the aggregation of benefits across individuals and for the application of the value transfer method. Each of the economic valuation methods outlined in section 2.3.2 is associated with conceptual and practical challenges. Even for market-based methods which are relatively more straightforward to apply; market prices may be distorted and adjustments may be needed to better reflect the economic value (Barbier et al., 1997).

The lack of markets and other direct behavioural links to underlying values and the diverse livelihood support and cultural roles of wetlands for many societies challenges the 'measurement' of wetland benefits in economic value terms (Heal et al., 2005; Turner et al., 2008). Wegner and Pascual (2011) warned of the likelihood of underestimating the value of ecosystem services crucial to livelihood sustainability due to the budget constraints of the financially poor. Further difficulties arise from knowledge failure. The economic valuation of wetland restoration is based on the preferences of individuals for restoring wetlands and maintaining or increasing the flow of services from them. It has been argued, however, that the public lacks information on the relationship between ecosystems and human well-being (e.g., Polasky and Sergerson, 2009; Costanza et al., 2017) and, further, that there are limits to the capacity of individuals to process such information and, therefore, to form rational preferences (Kahneman, 2003; Samson, 2016).

2.3.3.5 *Further considerations in practical application*

The economic valuation of wetland restoration necessitates integrating ecological and economic information. Fundamental to achieving this integration is at least some level of collaboration between natural scientists (e.g., wetland ecologists, hydrologists) and economists (and ideally social scientists as well), and information on both biophysical and social elements. The definitions and measures of ecosystem services must be similarly defined across both ecological and economic assessments (Heal et al., 2005). This is equally important in the design and implementation of monitoring programs to ensure that data collection is holistic and compatible across biophysical and social information needs. Regarding collaboration, Benda et al. (2002), while recognizing the necessity of interdisciplinarity and emphasising its potential to enhance problem solving, outlined several obstacles that regularly arise in collaborative efforts[46], which require additional time and effort to overcome.

Site specificity presents an additional challenge to the economic valuation of wetland restoration. Biophysical and social factors[47] affect wetland service supply, demand and the value of the associated benefits and these factors vary spatially (Woodward and Wui, 2001; Brander et al., 2006; Boyd and Banzhaf, 2007; Ghermandi et al., 2010; Chaikumbung et al., 2016; Mensah et al., 2017). This means that an understanding of the physical, social and political context within which the wetland restoration takes place is needed. Applying 'generic' per hectare values for wetland benefits (e.g., in a value transfer approach), while attractive as a low-cost practical tool to inform decision-making, ignores the spatial variability in both wetland service supply and demand and risks over- or under-estimating wetland benefit values (Polasky & Sergerson, 2009). The importance of context is amplified in the case of wetland restoration due to additional variation across sites from factors affecting the response of individual wetlands to restoration and their recovery time (Meli et al., 2014). Further, Morse-Jones et al. (2010) draw attention to spatial variation across beneficiaries, noting the 'distance decay effect' [48] and the potential for biased estimates if a constant unit value across a population is assumed.

A common practical challenge in the economic valuation of wetland restoration is the availability of site-specific information. The monitoring and evaluation of restoration

[46] These included: (i) the specificity of disciplinary 'language'; (ii) epistemological foundations embedded in a historical scientific and socio-political context which has changed; (iii) mismatches between disciplines in spatial and temporal scales of analysis, forms of knowledge and levels of accuracy and detail, and (iv) differing ontological assumptions (world views) across disciplines and the collaborators (individuals).
[47] For example, the biophysical characteristics of the wetland (type, slope, soil type), the size and wealth of adjacent and downstream populations, the surrounding and downstream land-use and economic activities, land tenure and governance arrangements, and availability and access to substitute services or benefits.
[48] The 'distance decay effect' is the case where utility declines as distance from the site increases (Morse-Jones et al., 2010).

outcomes is considered deficient, particularly over the medium to longer term (Ferraro and Pattanayak, 2006; Pendleton and Balderra, 2010; Wortley et al., 2013; Meli et al., 2014). Ideally, data should be collected across biophysical and social attributes and outcomes before and after the restoration intervention and for similar sites with no restoration (Ferraro and Pattanayak, 2006; Pendleton and Balderra, 2010). This level of monitoring and data collection can be practically challenging and expensive. Measuring and monitoring ecosystem services is further complicated by spatial differences between where services are generated and where they are used (and where behaviour can be observed) and the fact that some services, such as pollination, are not amenable (practically) to measurement (Fisher et al., 2008). Restoration cost data is also often incomplete (e.g., covering planning, establishment and maintenance costs) (Iftekhar et al., 2017). In the case of large, state funded restoration programmes (e.g., the South African Working for Wetlands programme) costs are generally recorded at the programme level rather than at the site scale.

Wetland systems are multifunctional (Turner et al., 2008) meaning that a unit of wetland can provide more than one service. In addition, a single service can be linked to several benefits. Wetland functions and processes are interrelated, and some wetland services may be complementary, whereas the relationship between others may be negative or even mutually exclusive (e.g., optimal water levels for agricultural cultivation and the carbon sink service are different). The challenge for economic valuation is the difficulty in treating ecosystem services and the related benefits as discrete items to avoid double counting[49], while also recognizing, and accounting for, their multiple benefits (de Groot et al., 2002; Boyd and Banzhaf, 2007).

There are a number of operational aspects that need to be considered in the economic valuation of wetland restoration. Attention to these elements is necessary to adhere to the principles of economic valuation (Bockstael et al., 2000) and to generate methodologically rigorous estimates of the economic value of wetland restoration. However, as stated by Freeman (2003:19) "The state of the art cannot be expected to advance to the point of

[49] Double counting can occur where: (i) an intermediate service is valued and subsequently valued again through its contribution to a final service or benefit; (ii) competing ecosystem services are valued separately and the values added together; or (iii) a service/benefit is valued more than once through the application of multiple valuation and the results are aggregated (Morse-Jones et al., 2010).

producing exact parameter values for all kinds of environmental change. This is because of the inherent uncertainty and imprecision in measurement techniques based on sampling and statistical inference and because of the complexity and imperfect understanding of the physical, biological, and socioeconomic systems that must be modelled to produce the relevant scenarios for welfare comparisons". For Parks and Gowdy (2013: e6), the "Theory [of economic valuation] has gone beyond the reality of data availability and basic conceptual problems with biophysical models of ecosystem services".

2.3.3.6 Societal goals and policy principles

It is the underlying goals to which a society aspires that define whether a particular outcome is favourable or not, whether it is 'valuable'. A primary rationale for conducting economic ecosystem valuation is to influence decision-making by introducing ecosystem benefits into CBA. This perspective accepts economic efficiency as a social goal and criterion for ranking decision options. However, economic efficiency is not unanimously accepted as the only, or primary, social goal. Much of the debate on economic ecosystem valuation questions whether the economic efficiency criterion is appropriate and sufficient to guide decisions on public goods and human well-being within the current context of ecological and social conditions (Daly, 1992; Costanza, 2000; O'Neill and Spash, 2000; Paavola and Bromley, 2002; Arrow et al., 2004; Farley, 2008; Vatn, 2009; Munda, 2016). Other societal goals, such as sustainable scale and just or fair distribution are regarded as equally or more imperative.

Scale refers to the "the physical volume of the throughput, the flow of matter-energy from the environment as low-entropy raw materials, and back to the environment as high-entropy wastes" (Daly, 1992:486). Sustainable scale relates to containing the scale of human activities within levels that support the long-term functioning of the global system (Costanza, 2000) and maintaining vital ecosystem functions for future generations (Ekins et al., 2008). Related to the consideration of sustainable scale are debates on the substitutability between forms of capital and the existence of critical natural capital (Ekins et al., 2003; Brand, 2009); the presence of ecological thresholds at which point the state of an ecosystem experiences fundamental changes (Farley, 2012); and limits to economic production and growth (Naess, 2006b).

Distribution refers to "the relative division of the resource flow, as embodied in final goods and services, among alternative people" (Daly, 1992:486). Fair distribution relates to equity considerations between humans, both within the current generation and across generations, and between humans and non-human species (Munda, 1997; Costanza, 2000). Distributional concerns also pertain to the distribution of environmental damages, for example, groups of people exposed to disproportionate amounts of pollution (Martinez-Alier, 2002) and spatial differences between those 'generating' environmental damages and those likely to suffer the consequences (Ekins et al., 2008).

Efficiency refers to the relative allocation of resources across alternative uses (Daly, 1992). 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). It is seldom the case that a reallocation of resources will not result in some individuals 'losing'. Instead, the compensation principle is applied whereby a reallocation of resources is considered desirable if the 'gainers' are able to compensate the 'losers' and still be better off (Young, 1996), that is, if the 'benefits' exceed the 'costs' in terms of individual utility satisfaction. Under the compensation principle, 'gainers' and 'losers' are treated 'symmetrically' and a loss to one individual can be off set against a gain to another. This assumes that individuals, regardless of their differences (e.g., in socio-economic status) experience a unit of loss or gain in the same way[50] and raises questions regarding fairness and equity.

Neoclassical economic valuation and CBA are tools for evaluating the contribution of an action (or object) towards the goal of economic efficiency. Given the assumptions inherent in the net present value and discounting methods, this type of evaluation favours benefits accrued in the short-term over long-term benefit flows (Alexander et al., 2016). The neoclassical economic framing does not necessarily lead to social fairness and sustainable scale (Costanza, 2000); stronger arguments suggest that it detracts from achieving such goals and is the 'cause'

[50] That is, the 'loss' of R100 to one individual is equivalent to the 'gain' of R100 to another, regardless of the proportion of that R100 in terms of their overall 'wealth'.

of many current social and ecological issues (e.g., Schneider et al., 2010; Wolff, 2013; Spash and Aslaksen, 2015).

Further to the limits of the neoclassical economics approach in addressing multiple social goals, Hettinger (2012) raised an interesting question in the context of ecological restoration – that is, whether restoration should be considered a 'net benefit' to society. Hettinger's argument stems from the perspective that ecological restoration actions are an attempt to make amends for a past wrong, an attempt to 'restore' what we have degraded. For Hettinger (2012:41), "restoration is a short-term and fundamentally regrettable way of relating to nature", a relationship that only makes sense "given the current and past abusive human treatment of nature". From this perspective, the long-term goal is "virtuous human flourishing" based on a respectful human-nature relationship where ecological restoration would have little role. For Hettinger (2012), echoing the sentiments of Katz (1992), addressing the cause of ecological degradation is the objective; ecological restoration is a compromise and should not be a basic policy goal.

2.3.4 Section conclusion

It is widely accepted that the vital role of ecosystems and biodiversity in maintaining human well-being has largely been ignored in land-use decision-making in mainstream westernized governance systems. Economic ecosystem valuation is perceived as a means to integrate ecological and economic information and considerations towards redressing the neglect of ecosystems in land-use decision-making. In this way, the contribution of ecosystem restoration is evaluated against the goal of economic efficiency. Economic valuation, along with CBA, is regarded as a useful tool to aid global wetland management.

This chapter has considered the rationale behind the economic evaluation of wetland restoration and reviewed the current state of knowledge on the economic valuation of wetland restoration as well as various challenges to, and critiques of, the standard neoclassical economic approach to evaluating wetland restoration. From the review and discussion presented in this chapter, the following conclusions can be made:

- Economic valuation methods impose a specific framing and rationality on the formation and development of values associated with ecosystems and ecological restoration;
- Views on the use and appropriateness of economic ecosystem valuation are diverse. Concerns stem from ideological and technical challenges associated with the assumptions and limitations of neoclassical economic theory, practical difficulties in application, and questions concerning the social goals that should guide policy and decision-making.

Whether the application of neoclassical economic valuation to ecosystem and biodiversity attributes is appropriate, and if, or how, the associated challenges are surmountable, continues to be much debated. Some suggest that economic ecosystem valuation should only be used in very specific situations framed by an appropriate valuation question, others view it as a transitory short-term tool in shifting the dominant economic paradigm, while others eschew its use altogether. Increasingly, ecosystem valuation is expanding beyond the standard neoclassical approach to incorporate insights and tools from other schools of economic thought and outside disciplines. If economic valuation is accepted as one tool in the evaluation of wetland restoration, then the challenge remains of how to ensure that economic valuation is used 'appropriately'; that it is integrated with other methods and information, and that it does not displace attention to the drivers of wetland degradation and considerations not amenable to an economic value calculus or does not obscure other interventions (e.g., institutional changes) towards sustainable wetland management.

Having reviewed key theoretical aspects underpinning the valuation of wetland restoration, the next chapter extends the discussion through a review of the application of economic valuation to wetland rehabilitation to gauge the extent to which the outcomes of wetland enhancement activities have been valued using economic methods and to identify trends and gaps within the applied research on the topic. Following the final review chapter, the thesis then turns to the case study discussion.

CHAPTER 3: THE ECONOMICS OF WETLAND RESTORATION: A QUANTITATIVE LITERATURE REVIEW

3.1 INTRODUCTION

The economics of ecosystem restoration is an emerging field of research with contributions from multiple disciplines and attempts towards inter- and trans-disciplinary research. These characteristics can make it challenging to gauge the extent of existing research through a 'traditional' narrative review. As an alternative, Pickering and Byrne (2014) proposed a 'systematic quantitative' approach to review the literature that, they argue, is particularly suited to new fields and transdisciplinary research. The approach provides a means of mapping the breadth of literature on a given topic and identifying trends and gaps within the literature base (Pickering & Byrne 2014).

The approach of Pickering and Byrne (2014) was applied to review the international published literature on the economic valuation of the active intervention in a wetland system to alter or maintain its condition[51]. The review process was used to address several questions: (1) to what extent have the outcomes or benefits of wetland enhancement activities been valued using economic methods; (2) what are the temporal and spatial trends in publications on this topic; (3) for what purpose have such assessments been undertaken; and (4) which methods have been applied and attributes (benefits and costs) studied?

Several earlier literature reviews have been undertaken in related fields: three meta-analyses of wetland values (Woodward and Wui, 2001; Brander et al., 2006; Ghermandi et al., 2010), five reviews related to the economics of ecological restoration (Aronson et al., 2010; Robbins and Daniels, 2012; Blignaut et al., 2013, de Groot et al., 2013 and Blignaut et al., 2014) and four reviews on evaluating ecological restoration (Ruiz-Jaen and Aide, 2005; Ntshotsho et al., 2011; Wortley et al., 2013; Meli et al., 2014). None of these reviews have focused on the economic valuation of wetland restoration outcomes.

---

[51] The material in this chapter is based on an article published as BROWNE, M., FRASER, G. AND SNOWBALL, J., 2018. Economic evaluation of wetland restoration: a systematic review of the literature. *Restoration Ecology*, 26 (6), pp. 1120-1126. (Research conceptualised, analysed and written up by M Browne, with advice and assistance from G. Fraser and J Snowball in their supervisory capacity).

3.2 METHODS

A systematic quantitative literature review was performed following the approach outlined in Pickering and Byrne (2014) and illustrated in Figure 3.1.

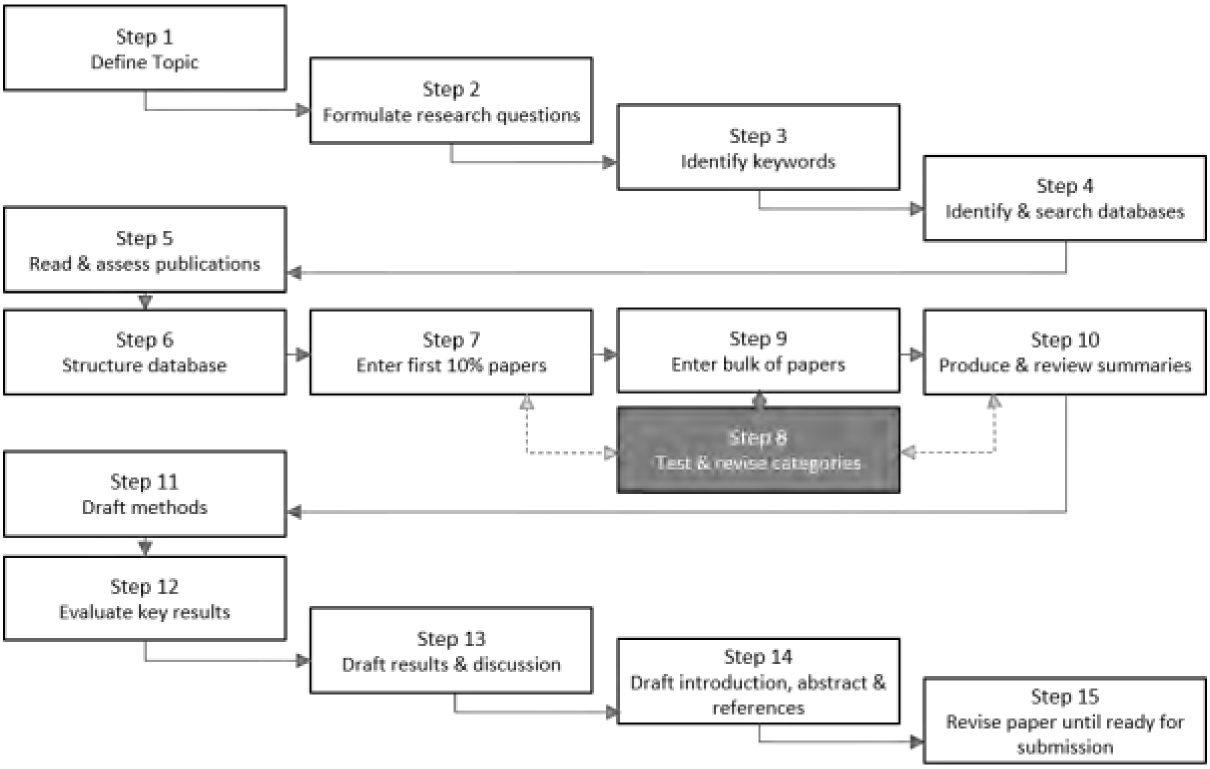


Figure 3.1: The Pickering and Byrne (2014) quantitative literature review process.
Source: Reproduced from Pickering and Byrne (2014:539).

Existing literature reviews in the field of ecosystem restoration were used to inform the selection of search terms and identify appropriate search platforms (electronic databases). The key characteristics noted from the reviews are summarized in Appendix 3.1. Search terms were selected to distinguish studies relating specifically to the economic valuation of wetland enhancement activities. Numerous searches using various terms and groupings of terms were undertaken in a 'trial and error' approach. The final search comprised of the set ((wetland* OR marsh*) AND (restor* OR rehabilit* OR creation OR interve*) AND (valuat* OR "cost benefit" OR "benefit cost" OR "economic analysis" OR "economic assessment" OR "willingness to pay")). A search of the title, abstract and keywords of journal articles across the Web of Science (Thomson Reuters), Science Direct (Elsevier) and EBSCOhost electronic databases was performed. The EBSCOhost database was included to cover the journals *Ecological Management and Restoration* and *Ecological Restoration* which are not covered by

Science Direct or Web of Science. The search was limited to English language peer-reviewed journal publications. Books, book chapters and grey literature were not considered in the review. The search was conducted for publications across the earliest dates available within the electronic databases to January 2016 (inclusive).

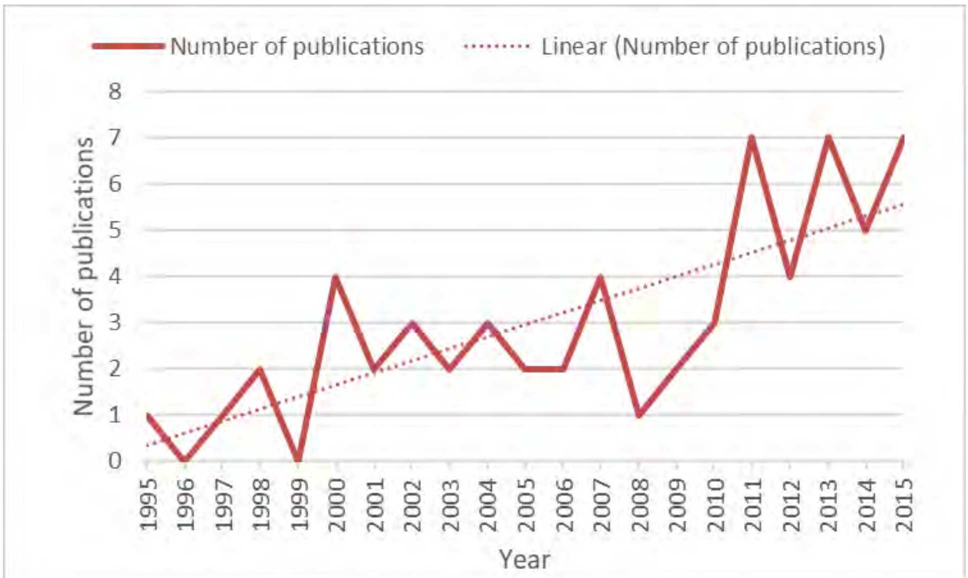
All publications returned by the search were subjected to a preliminary 'screening of abstracts' to ensure they met with the specific criteria of (1) including an applied economic or value analysis (i.e., not only a framework, review or discussion of values), (2) the study related directly to an active intervention in a wetland system, and (3) the publication was an original research paper (i.e., peer-reviewed and of primary source). Only publications fulfilling these criteria were retained for further review. For each publication, information pertaining to three broad categories, each including multiple sub-categories, was recorded in a Microsoft Excel database: (1) publication and study details (5 sub-categories), (2) details related to the economic analysis (31 sub-categorise), and (3) details of the wetland enhancement activity (21 sub-categories). The database was used to summarise the key elements of the publications in answer to the proposed research questions.

3.3 RESULTS

A total of 365 publications were retrieved, of which 63 met the required criteria and were retained for entry into the database. Publications spanned 42 journals covering a range of disciplines including economics, engineering, ecology, conservation and environmental management. The greatest number of publications in a single journal appeared in *Ecological Economics*, nine papers (14%), followed by *Ecological Engineering* with four papers (6%). Several journals were represented by two or three publications and 33 journals (52%) were represented by a single publication. The discipline of economics was represented in 14 publications (22%); nine papers in *Ecological Economics*, two in *Marine Resource Economics* and one publication each in the *American Journal of Agricultural Economics*, the *Canadian Journal of Agricultural Economics* and *Land Economics*. Given the inter-disciplinary or multi-disciplinary nature of many of the publications, the affiliation of the authors was examined using the author information provided in the publication. For 31 (49%) of the publications, at least one of the authors was affiliated with an economics department or institution. For 27

(43%) of the publications none of the authors appeared to be affiliated with an economics related department or institution. The author affiliation was unclear in five publications. Of the 31 studies involving an economics department or institution, 10 (16%) studies appeared to be undertaken solely by authors affiliated with an economics department or institution. Looked at another way, 21 (33%) of the publications were a collaboration between economists and non-economists (as generalized from author affiliation). Examination of the frequency of author contributions across the 63 publications found that 11 authors (of a total of 211 authors) contributed to more than one publication: 10 authors contributed to two publications each and one author appeared on three publications. Seven of the 'repeat contributors' related to the same four publications.

The trend in the number of publications over time is shown in Figure 3.2. The earliest publication appeared in the *Journal of Soil and Water Conservation* in 1995 and investigated the restoration of farmed wetlands in the United States Corn Belt. Since then, publications of the economic valuation of wetland enhancement activities have increased from four publications in the 1990s to 25 publications during the period 2000 to 2009. An additional 34 papers (54% of the total) were published during the period 2010 to January 2016 (inclusive).



Note: The dotted line shows the linear trend in publications over time.

Figure 3.2: The temporal trend in the publications examined in the literature review (N=63 studies).

The geographic distribution of the publications is presented in Figure 3.3. The wetland enhancement projects were predominantly located in North America (59%) followed by Europe (24%). There was a single study from Africa (Kenya) and none of the studies originated from South America. The earliest publication of a study outside of North America was in Europe, in 1997.

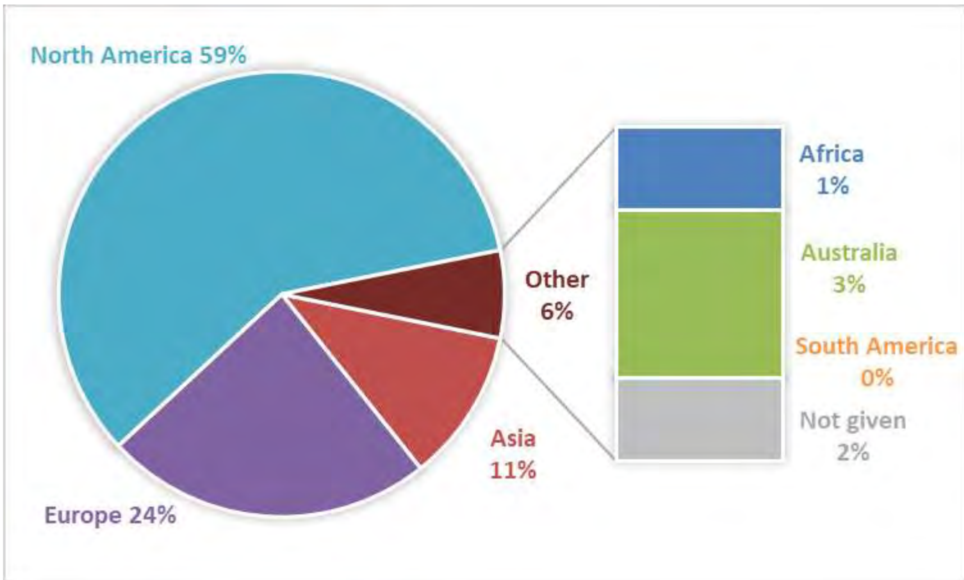


Figure 3.3: The geographic spread of the publications examined in the literature review (N=63 studies).

The characteristics of the economic analyses reported in the 63 publications were reviewed. Firstly, the timing of the economic analysis in relation to the stage of the wetland enhancement activity was examined. In 38 of the studies (60%), the economic analysis was conducted prior to the completion of the wetland enhancement (pre-intervention); of which 19 examined the case of a proposed enhancement activity, 16 considered a more general or hypothetical situation and three were undertaken during intervention activities. In 21 of the studies (33%), the economic analysis was performed after the wetland intervention had occurred (post-intervention). For 10 of these studies the economic analysis was undertaken within 10 years of the intervention, two studies included sites 'older' than 10 years, but within 20 years, and nine publications did not specify the number of years. Three studies included sites at both pre- and post- intervention phases; and one study did not specify the timing. The timing of the economic analysis in relation to the stage of the intervention across the 63 publications is summarized in Figure 3.4.

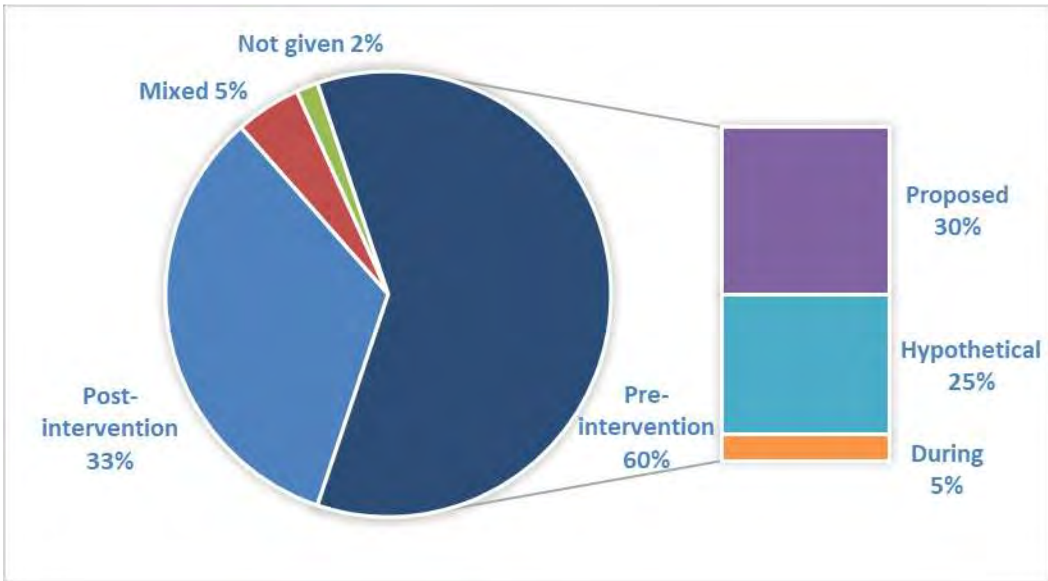


Figure 3.4: The timing of the economic analysis in relation to the stage of the wetland enhancement activity across the publications examined in the literature review (N=63 studies).

For the 63 publications, the intended purpose of the economic analysis was identified and grouped into nine categories. The results are summarized in Table 3.1. In six of the publications more than one purpose was indicated. The most frequently cited intention, 19 publications (30%), was to 'contribute to learning', which included intentions to 'inform best practice', 'develop or improve evaluation methods', and 'inform decision-making and management in general'. The intended purpose of 10 studies (16%) was to inform a specific decision and the analyses were generally undertaken before any wetland intervention occurred (one study included sites at both pre- and post- intervention stages and one study did not specify the timing). Of the four studies which indicated the intended purpose of evaluating project performance or success, three were applied after the intervention was completed and one during the intervention. Nine studies (14%) were conducted for research purposes, generally to investigate a particular methodological query. Two publications, both studies of wetlands in China, indicated that the intended purpose of the economic analysis was to inform or identify financing options for wetland enhancement activities. One study was conducted prior to the intervention to determine amounts for a payment for ecosystem services programme and one was undertaken after the intervention to identify future opportunities or sources to raise finance for wetland protection and to develop mechanisms to reward (and penalize) good (poor) management practices.

Table 3.1: The intended purpose of the economic analysis of wetland enhancement activities reported in the publications examined in the literature review (N=63 studies)

Intended purpose	**No. of studies**	**Pre-intervention**	**Post-intervention**
Contribute to learning	[a]19	10	8
Inform a specific decision	[b]10	8	0
Research	9	5	4
Provide evidence of benefits/value created	8	4	4
Assess public support for enhancement	7	7	0
Determine feasibility	[c]6	4	1
Evaluate project performance	4	[d]1	3
Inform financing options	2	1	1
Not specified	4	2	2

Notes: Pre- and post –intervention refer to the timing of the analysis, before or after the wetland intervention was completed, respectively; several studies indicated more than one purpose. [a] One study included sites at both stages; [b] One study included sites at both stages and one study did not specify the time; [c] One study included sites at both stages; [d] the evaluation was undertaken during the intervention, before it was completed.

Three types of economic analysis were applied to assess the wetland enhancement activities: benefit valuation, cost analysis and economic impact assessment. Of the 63 studies reviewed, 50 applied benefit valuation, of which two also included an economic impact analysis component. Several benefit valuation studies also considered the costs of the enhancement. Of the benefit valuation studies, 29 (58%) were applied before the intervention was completed and 18 (36%) after the intervention (2 studies included sites at both phases and for one study the timing was not specified). In 13 of the studies, only the costs of the enhancement were estimated: nine as a pre-intervention assessment, three as a post-intervention assessment and one study included sites at both phases.

The methods applied across the 50 benefit valuation studies are summarized in Table 3.2, several of the studies applied more than one method. The maximum number of methods employed in a single study was three, six studies applied three methods, nine studies applied two methods and 35 studies applied one method. Across the 50 benefit valuation studies, stated preference methods were used most often, followed by cost-based methods. Stated preference methods – including contingent valuation and choice modelling approaches - were applied 22 times. In 18 instances, stated preference methods were applied before any wetland intervention was undertaken, in four cases the analysis was applied after the intervention was completed. 'Willingness to Pay (WTP)' was estimated in 20 of the 22 applications, two studies estimated 'Willingness to Accept (WTA)' (of which one also estimated WTP), and one study did not estimate monetary values. Cost-based methods were applied 15 times, 10 of which were performed prior to the wetland intervention and five after the intervention. The replacement cost approach was the main cost-based method used, it was applied in nine cases, six in a pre-intervention context and three times post-intervention.

Table 3.2: Methods used in the valuation of wetland enhancement activities in the studies examined in the literature review (N=50 studies)

Method	**Number of times applied**		
	Total	**Pre-intervention**	**Post-intervention**
Stated preference	**22**	**18**	**4**
Choice modelling	11	10	1
Contingent valuation	10	8	2
Not specified	1	0	1
Cost-based	**15**	**10**	**5**
Replacement cost	9	6	3
Avoided cost	2	2	2
Avoided damage	4	2	2
Value transfer	**12**	**7**	**5**
Value transfer	10	6	4
Value-function transfer	2	1	1
Market-based	**11**	**4**	**7**
Market prices	11	4	7
Revealed preference	**3**	**0**	**3**
Travel cost	1	0	1
Hedonic pricing	1	0	1
Expenditure analysis	1	0	1
Optimization	[a]**3**	**1**	**1**
Other (non-monetary)	**3**		
Emergy analysis	1	[b](Mixed)	
Habitat value	1	[b](Mixed)	
Ranking of importance	1		1

The wetland enhancement attributes evaluated across the 63 studies were examined and are summarised in Table 3.3. Of the 63 publications reviewed, 33 studies evaluated a single attribute, 24 studies assessed two to five attributes and six studies considered more than five attributes, with a maximum of nine attributes assessed in a single study (using a non-monetary valuation approach). Habitat provision was the most common attribute (25 papers, 40%) and was predominantly assessed using the stated preference method (13 studies); a cost-effectiveness approach was adopted in five of the studies. The category 'habitat provision' included habitat provided for a particular species (most commonly waterfowl habitat), for biodiversity support and for a general increase in wetland habitat.

Table 3.3: Wetland enhancement attributes evaluated in the studies examined in the literature review (N=63 studies)

Attribute	No. of studies	Percent of studies
Habitat provision	25	40
Water quality	19	30
Recreation	15	24
[a] Material provision	9	14
Fishery production	8	13
Carbon sequestration	7	11
Flood prevention	7	11
Gas regulation	7	11
Combined benefit	6	10
Water supply	5	8
Amenity	4	6
Water regulation	4	6
Erosion control	4	6
Storm protection	4	6
Biodiversity	3	5
Income	3	5
Preferences for management type	2	3
Property value	2	3
Costs only	2	3
Climate regulation	2	3
Cultural	2	3
Education	2	3
Disturbance regulation	1	2
Hydrological characteristics	1	2
Reduction in maintenance costs	1	2
Species characteristics	1	2
Top soil	1	2

Notes: [a] Includes the categories of food, fibre, agriculture and un-specified provisioning benefits.

Water quality benefits were evaluated in 19 studies (30%), while water supply and water regulation benefits were considered in five and four studies respectively. Water-related benefits were primarily valued using cost-based approaches, specifically the replacement cost method, followed by the value transfer method. Cost-effectiveness, stated preference, market-based and non-monetary approaches were the other methods used to evaluate water-related benefits of wetland enhancement activities. The recreation-related benefits of wetland enhancement activities were assessed in 15 studies (24%) and valued mainly using stated preference (7 studies) and value transfer (4 studies) methods. Altogether, 16 studies (25%) evaluated climate-related services or benefits (including carbon sequestration, gas regulation and climate regulation). Cost-based methods were applied in five of the climate-related benefit valuations and stated preference, value transfer and market-based methods were each used three times. The cultural benefits of wetland enhancement activities were assessed in two studies, one using the stated preference method and the other a non-monetary approach. In six of the studies, the benefits of wetland enhancement were valued as a combined benefit (i.e., not disaggregated into separate attributes) using stated preference, value transfer and optimization methods (2 studies each). The income impacts of wetland enhancement activities were considered in three studies and assessed using an economic impact approach (2 studies) and the stated-preference method (1 study).

Whether the costs and benefits of the wetland enhancement activities were compared - in those studies that estimated both - was examined. Of the 63 studies reviewed, 50 included estimates of the benefits of wetland enhancement activities and 40 included cost estimates. The costs and benefits were compared in 26 studies. The discount rates and timeframes applied in the studies were also examined. A three percent discount rate was used in four of the studies, a four percent and five percent rate was used in three studies each and one study applied a nine percent discount rate. Seven of the 26 cost-benefit comparisons employed a range of discount rates and eight studies did not use or specify a discount rate. The timeframes used in the cost-benefit comparisons are summarised in Figure 3.5. In seven of the studies, a range of timeframes was considered, and for six studies a timeframe was either not given or not used in the comparison. A comparison of discount rates and timeframes showed no obvious relationship between the two (i.e., there was no corresponding trend in the timeframe used as the discount rate increased). However, the comparison was limited to a small sample size: only those studies that reported both a discount rate and timeframe (10 studies). A scatter plot of timeframe against discount rate is given in Figure 3.6.

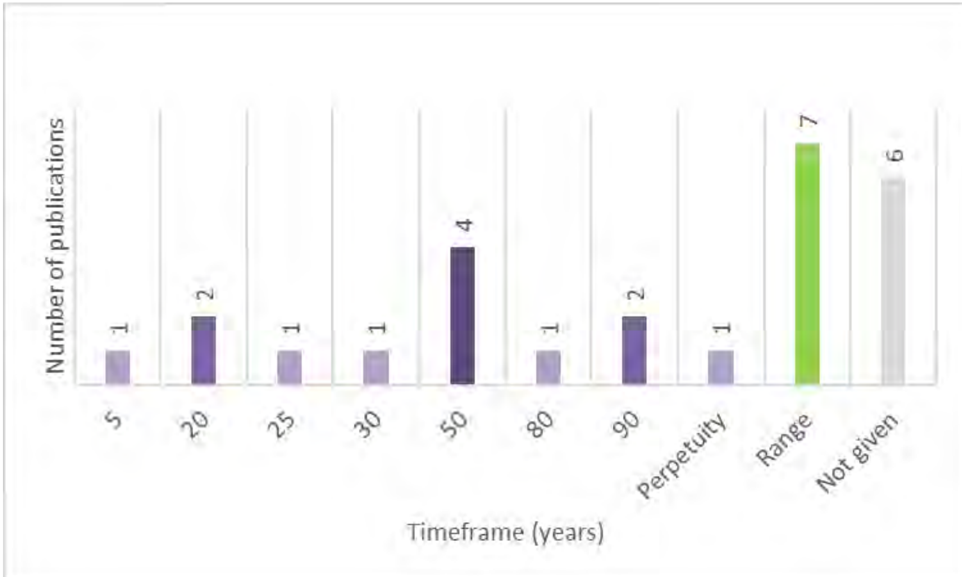
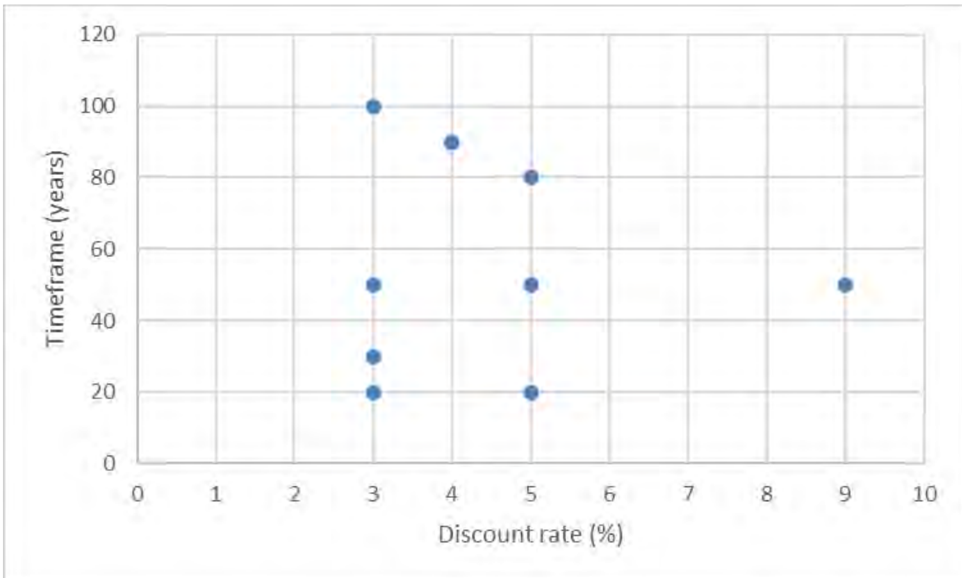


Figure 3.5: Timeframes used in the comparison of costs and benefits of wetland enhancement activities examined in the literature review (N=26 studies).



Notes: Two studies applied a four percent discount rate and a 90 year time frame (i.e., only nine of the ten points are visible in the plot).

Figure 3.6: Scatter plot of timeframes against discount rates used in the comparison of costs and benefits of wetland enhancement activities examined in the literature review (N=10).

Of the 26 studies that estimated both costs and benefits, 16 reported benefit cost (B-C) ratios, either directly or in such a way that B-C ratios could be easily calculated. Eleven of the 16 studies estimated a range of B-C ratios based on several scenarios. The lowest B-C ratio reported for wetland enhancement activities was zero (i.e., no benefits relative to costs) and the highest was eight. For 10 studies, at least one of the B-C ratios estimated was greater than one: for six studies, the highest B-C ratio estimated was below one. A discounted payback approach and an internal rate of return estimate were used in two studies (one study each).

In addition to considering the economic analyses, several characteristics of the enhancement interventions were examined, including (a) the entity responsible for initiating the intervention, (b) the intended purpose of the intervention, (c) the funding strategy for the intervention, and (d) land ownership. In two instances, two different publications were based on the same enhancement activity; there were 61 different enhancement activities across the 63 publications examined. Accordingly, the sample size for the following discussion is 61 and not 63.

In terms of the entity responsible for initiating the rehabilitation, the category 'government' dominated (46% of studies). All enhancement activities undertaken for 'compliance with regulation' purposes were regarded as being initiated by a government entity. In 25 percent of the publications, the entity responsible for initiating the enhancement activity was not clear. In eight of the publications, the enhancement activity was initiated by either a conservation, a non-profit or a non-governmental organisation. 'Citizens' and 'private company' were responsible for initiating the enhancement activity in one study each, while research institutes were the responsible entity in three of the studies.

Six broad categories of the 'intended purpose of the intervention' were apparent; the most dominant was 'public service' (39%), followed by 'conservation' (23%). The results are summarized in Figure 3.7. One of the conservation studies specifically highlighted 'support fishery production' as a second intention, effectively representing public and private sector motivations. In nine studies (15%), the enhancement activity was undertaken to comply with regulation requirements. 'Research' purposes were reported in two studies and one study each indicated 'private service' and 'protection of cultural heritage' as intended purposes. In eight publications the intended purposes of the enhancement activity was not clear and two studies reported multiple purposes.

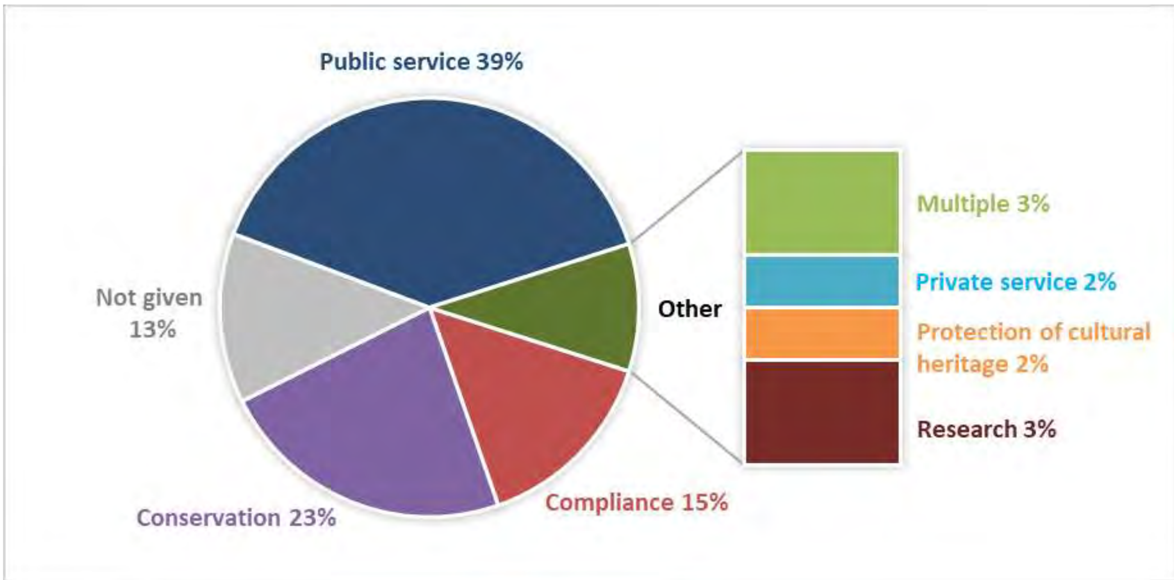


Figure 3.7: Intended purpose of the wetland enhancement activities examined in the literature review (N=61 studies).

In 39 percent of the publications, the source of funding for the wetland enhancement activity (funding strategy) was not given. For those publications indicating the funding strategy, use of 'public funds' was predominant (38%). 'Research funds' were used in four instances: in two cases the indicated intended purpose was 'research', in one instance it was to provide a 'public service' (and 'government' was given as the responsible entity) and in one case it was for 'conservation' purposes.

The ownership of the land on which the enhancement activity was undertaken was also examined. Unfortunately, this characteristic was infrequently reported in the publications: in 61 percent of the publications land ownership was not clear. Ten studies indicated a mix of 'government' (public) and 'private' land ownership. This occurred when the intervention covered a large area with different land owners, or when the study considered multiple sites with different land ownership. Eight studies reported 'private' ownership, while four indicated 'government' ownership. Land ownership by a non-profit or non-governmental organisation was indicated in two studies.

3.4 DISCUSSION

The quantitative literature review was undertaken to investigate (1) the extent of research on the economic valuation of wetland enhancement; (2) the temporal and spatial trends in publications on this topic; (3) the intended purpose of the studies on this topic; and (4) the methods applied and attributes (benefits and costs) studied.

3.4.1 Findings

In answer to the first question, 63 publications met the criteria of an original economic analysis of the active intervention to enhance a wetland ecosystem, 21 of which were conducted after the completion of the intervention. Since there are no other reviews of the same topic, which the author is aware of, it is difficult to draw comparisons about the number of publications identified. A meta-analysis of the values of natural and human-made wetlands identified a similar number of journal publications (68); only 14 of which assessed the economic values of created or restored wetlands, but all were published prior to 2010 (Ghermandi et al., 2010). In a 2006 review of wetland valuation studies (i.e., not specific to wetland enhancement), 191 studies were identified, sourced from both journal and non-journal material and including review and discussion articles; 80 of the studies were considered suitable for use in a meta-analysis (Brander et al., 2006). In a broad search using the terms (“restore” or “restoration”) and (“economy” or “economics”) and (“ecology” or “environment” or “ecosystem”), Blignaut et al. (2014) identified 650 studies including reviews and did not distinguish applied economic assessments from general discussions of the economics of ecological restoration. Robbins and Daniels (2012) identified 97 studies of the valuation of terrestrial restoration projects in the United States during the 2000 to 2010 period. Wortley et al. (2013) identified 301 publications dealing with the post-implementation evaluation (not restricted to economic evaluation) of terrestrial restoration projects, but found “no economic measure of ecosystem services in post-implementation evaluations” (Wortley et al., 2013:541). Given the strict criteria imposed in this review (applied economic analysis of active intervention in a wetland ecosystem), the broad range in numbers of studies identified in related reviews, and that the field of restoration economics is considered relatively new (Blignaut et al., 2014), the number of publications identified in this review appears acceptable.

Temporal trends show an increase in publications on the topic over time, with more than half the publications appearing since 2010. Similar trends were reported in the review of the 'economics of restoration' by Blignaut et al. (2014) as well as the review by Wortley et al. (2013) on ecological restoration. The notable increase in publications after 2010 is attributed to the publication of the Millennium Ecosystem Assessment and the first major TEEB (The Economics of Ecosystems and Biodiversity) report in 2010 (Blignaut et al., 2014).

The geographic distribution of the publications shows that the research is heavily skewed toward North America and Europe. Similarly, the review of 'restoration economics' by Blignaut et al. (2014) and the terrestrial ecological restoration review by Wortley et al. (2013) found that the majority of research originated from projects in North America. In the wetland valuation reviews of both Brander et al. (2006) and Ghermandi et al. (2010), North America was the most studied region. In a slightly different interpretation, Aronson et al. (2010) reported 78 percent of the publications in their analysis of the literature on ecological restoration to be based on projects in high income countries. A single study from Africa was identified in this review and no studies originating from projects in South Africa were identified. In contrast, both the wetland valuation reviews of Brander et al. (2006) and Ghermandi et al. (2010) reported studies originating from wetlands in Africa (53 of 418 studies in the case of Ghermandi et al., 2010). However, both the wetland valuation reviews included literature from sources outside of published journal articles, in contrast to this review which was limited to journal publications. The under-representation of Africa is consistent with the findings of Wortley et al. (2013) and Aronson et al. (2010) in their reviews of terrestrial ecological restoration; both their studies were also restricted to journal articles. The under-representation of Africa observed in the present review may point to (1) an absence of wetland enhancement projects in Africa; (2) limited economic valuation studies of wetland enhancement activities in Africa; or (3) a gap in the published research on wetland enhancement activities in Africa.

The studies were published across a wide range of journals, indicating a diverse cross-disciplinary interest in the economic assessment and valuation of wetland restoration outcomes. Based on author affiliation, a collaboration between disciplines (economists and non-economists) was evident in a third of the publications, whereas almost half of the studies appear not to have included an economic department or institution. While the overall number of publications on the topic has increased overtime, the number of collaborations does not show as strong a trend: during the 1995 to 1999 period 75 percent of publications on the topic were a collaboration, during the 2000 to 2009 period 32 percent of the publications were a collaboration. For the current period, 2010 to 2016, 29 percent of publications were a collaboration. Given the relatively short time span (20 years), firm conclusions cannot be made about these observations. Furthermore, these observations were based on the classification of 'economist' from the affiliation of the authors and may not represent the actual disciplinary background of the author. For example, an author could be an economist based at an ecological institute, the author affiliation would then appear as a 'non-economist'. The apparent trend of a decline in collaborative publications could rather be a reflection of an increase in multi-disciplinarity within organisations themselves. A closer examination of author affiliations and collaborations across disciplines may be useful in future reviews. An examination of the frequency of author contributions across the 63 publications found no indication of a leading author or group of authors on the topic.

A further research objective was to examine the intended purpose of the economic analyses. Eight of the studies examined in this review indicated an intended purpose of providing evidence of the value created by the enhancement activity, of which only four were performed after the intervention was completed. This finding, along with the relatively lower proportion of post-intervention valuation studies, lends support to previous claims in the literature of a gap in the knowledge base regarding the realized (post-intervention) value of ecological restoration (van Zyl et al., 2004; Rey-Benayas et al., 2009; Blignaut et al., 2013; Wortley et al., 2013). However, more than half of the post-intervention valuation studies were published after 2009 with the most published in a single year in 2015, indicating that the number of studies on the realized value of wetland enhancement is increasing.

The final research objective was to determine which economic methods have been applied in the valuation of wetland enhancement and which attributes have been studied. Of the 63 studies reviewed, 50 applied benefit valuation methods of which 18 (36%) were conducted after the enhancement activity was completed. Even for the post-intervention analyses, it was not always the case (or clear) that the study measured the realized benefit of the wetland enhancement rather than the expected or projected benefits. The results suggest that less research attention has been given to the realized value of wetland enhancement activities compared to the estimated or projected value. Similar conclusions were drawn in two previous reviews of the literature on ecological restoration (Aronson et al., 2010; Wortley et al., 2013).

Stated preference methods were applied most often and to a wide range of attributes, but more frequently to value habitat provision and recreation characteristics. This is in contrast to the findings of the wetland valuation review of Brander et al. (2006) where the market-price approach was dominant. Several contributing reasons for the discrepancy are noted. Stated-preference techniques are generally the most expensive of the standard economic valuation methods to apply which is likely to influence their use. The Brander et al. (2006) review considered a larger set of publications (191) across a broader research focus (wetland valuation), whereas the present review considered a narrower and emerging research focus (wetland enhancement valuation), which may explain the more frequent use of stated-preference techniques observed in the present review. Given the elapsed time between the two studies (ten years) a change in the methodological focus of economic valuation research may explain the less frequent use of the market-based approach observed in the present review. Market-based methods of valuation have been applied extensively, as such 'new' publications on their application are likely to be lower than studies applying less developed valuation techniques such as stated-preference methods. The result may also indicate a growing emphasis of the non-market related benefits and values of (wetland) ecosystems and ecosystem enhancement and, as a result, an increased focus on developing non-market valuation methods, such as stated-preference approaches.

In this review, applications of the market-based method were less frequent, but the most commonly used method to value the provision of materials such as food and fibre, which makes theoretical sense, as often there is an existing market for such materials. After stated preference methods, the cost-based approach was the predominant method applied in valuing the benefits of wetland enhancement. Strictly, values estimated using the cost-based approach are not a true reflection of economic value as the estimates are not based on individual preferences and do not measure an individual's willingness to pay for the benefit (Heal et al., 2005). Cost-based estimates tend to reflect a lower-bound of the value of the benefit (Brouwer et al., 2013). Consistent with the findings of Brander et al. (2006), revealed preference methods were used to value amenity and recreation benefits, but were least frequently applied.

A comparison was made between studies undertaken before (pre-intervention) and after (post-intervention) wetland enhancement activities were implemented. Stated-preference and cost-based methods were the most commonly applied methods in pre-intervention assessments. In post-intervention assessments, market-based, value transfer and cost-based methods were most common. Given that post-intervention studies appear to be performed less frequently than pre-intervention studies and the relatively common use of the value transfer approach in post-intervention studies, the question arises of where 'values' have been transferred from.

More than half of the studies examined considered only a single attribute, a significant number valued two to five attributes and a small portion considered more than five attributes. A similar trend was reported by Brander et al. (2006). The finding indicates a tendency towards partial valuation rather than a rigorous attempt to estimate the TEV of wetland enhancement activities. Habitat provision was the most studied attribute. Similarly, Brander et al. (2006) found the 'habitat and nursery' service to be the most commonly assessed, particularly as support for commercial fisheries and hunting. It is interesting to note that both this literature review on the value of wetland enhancement benefits and the review by Brander et al. (2006) on the valuation of wetlands, found habitat provision to be the most commonly studied benefit; yet the two reviews differ in the finding of the most frequently applied method. In the Brander et al. (2006) review the market-price approach was dominant. In this review, none of the habitat provision studies applied a market-price approach; stated-preference methods were predominantly used in valuing a change in habitat provision. The reason for this divergence may be related to the classification of 'habitat provision' by the two reviews. It may also reflect a temporal change in the use of valuation methods since the Brander et al. (2006) review predates this one by 10 years. As previously discussed, market-based methods are now considered relatively standard valuation methods, whereas stated-preference techniques are still developing. Furthermore, several influential reports addressing the valuation of ecosystems and drawing attention to the range of valuation methods have been published since 2006, including including reports from the TEEB project and reports from the United Kingdom Department for Environment, Food and Rural Affairs (DEFRA, 2011) and the United States Environmental Protection Agency (EPA, 2009).

The next most frequently assessed attributes reported by Brander et al. (2006) were amenity, recreational hunting and recreational fishing. These wetland services are generally traded in the market and have market-related values, as reflected in the relatively common use of the market-based valuation found in the Brander et al. (2006) review. The results of this review - of the specific case of wetland enhancement – deviate somewhat, in that the next most commonly considered attribute was water quality which links to the more frequent use of cost-based methods found in this study compared to the Brander et al. (2006) review. These findings highlight that valuation methods are linked to the attributes or benefits under consideration. In the context of wetland enhancement valuation studies, the greater prevalence of cost-based methods is likely related to the restoration and creation of wetlands for the specific purpose of water quality enhancement; cost-based methods being the predominant method used to value changes in water quality.

While the focus of this review was not on the costs of wetland enhancement activities, several earlier literature reviews suggest that the costs of restoration are less frequently reported or analysed, compared to the benefits (Robbins and Daniels, 2012; de Groot et al., 2013; Meli et al., 2014). Of the 63 studies examined in this review, 40 considered the costs of wetland enhancement activities, of which 14 were post-intervention assessments. In comparison, 50 benefit valuation studies were identified, of which 18 were performed post-intervention. In the context of this review, greater research attention has been given to estimating the costs of wetland enhancement activities before any intervention in the wetland, than has been given to recording and analysing the actual costs post-intervention. However, given the focus of the search terms (valuat* OR "cost benefit" OR "benefit cost" OR "economic analysis" OR "economic assessment" OR "willingness to pay"), it is likely that a significant portion of ‘cost only’ assessments were not identified in this search.

Cost-effectiveness and cost-benefit analyses were the two main evaluation or decision support tools employed in the studies examined in this review. The benefits of wetland enhancement activities were compared to the costs in approximately half of the benefit valuation studies, with a similar frequency of application in both pre- and post-intervention studies. While a multi-criteria analysis approach has been suggested as a possible tool for decision-making in the context of wetland management (Turner et al., 2000), none of the publications examined in this review applied a multi-criteria analysis approach.

One of the challenges of valuing ecosystem benefits is the selection of an appropriate discount rate to determine the present value of future ecosystem benefits. Discounting was not applied (or reported) in all of the studies examined in this review; of those that did, a greater proportion used a range of discount rates rather a single rate. The use of a range of discount rates is a reflection of the on-going debate over an appropriate discount rate for ecosystem valuation, as well as an indication of the potential sensitivity of benefit-cost comparisons to the choice of discount rate.

For those studies that reported benefit-cost ratios for wetland enhancement projects, estimates ranged from a low of zero (i.e., no benefits relative to costs) to a high of eight. In an analysis of the costs and benefits of restoration activities across multiple ecosystem types, de Groot et al. (2013) reported a range of benefit-cost ratios for wetland restoration of 0.05 to 11. The range of values reported across the publications examined in this study are consistent with the findings of de Groot et al. (2013). However, care should be taken in comparing the benefit-cost ratios, as the benefits and costs assessed and methods applied varied widely across the studies.

In terms of the characteristics of the wetland enhancement projects, a ‘typical’ wetland enhancement intervention was initiated by a government entity to provide a public service or for conservation purposes. Compliance with regulation was the next most frequently cited purpose. Enhancement activities were most commonly funded through public funds, although more than a third of the publications examined did not report the funding strategy used. Similarly, land ownership details were under-reported (61% of studies). A mixed (a combination of private and public or government) land ownership was most commonly reported (16%) followed by private land ownership (13%). In comparison, Aronson et al. (2010) found that the majority of restoration activities examined in their study were conducted by people based at research institutions (66%) followed by government agency-based researchers (19%). The likely reason for the discrepancy in findings (on the entity responsible for initiating the intervention) is the different focus of the two reviews. The Aronson et al. (2010) review focused on published research on the restoration of ecosystems, and was likely biased towards restoration activities undertaken for research purposes. The present review examined research on the economic analysis of enhancement activities (and not research on the enhancement activities themselves). However, the findings of the present review are likely biased towards enhancement activities undertaken by government entities who are more financially able to pay for economic assessments and are perhaps more motivated to undertake such studies as a justification for the spending of public funds. In addition, the Aronson et al. (2010) review predates this one (by more than 5 years). The finding may suggest that the enhancement of wetland ecosystems has become more widely accepted (and implemented) outside of the research arena since 2010, or that wetland enhancement is more accepted (and implemented) outside of the research arena compared to the enhancement of other ecosystems. A comprehensive review of ecosystem enhancement projects (including the grey literature) would be needed to confirm or reject this suggestion.

3.4.2 Limitations

Several limitations of the review are acknowledged. Firstly, the selection of publications was biased towards English language publications, other language papers were excluded, which is likely to have influenced the geographic distribution of publications and research on the topic. The search was restricted to journal publications and excluded all grey literature such as book chapters and reports. Similarly, this may have affected the results of the geographic distribution of publications, and influenced the trends observed in methods used and attributes studied. In the South African context, no publications originating from wetland enhancement projects in South Africa were identified, yet extensive wetland restoration has occurred (Phillips and Madlokazi, 2011; Kotze and Ellery, 2008). This finding could indicate a lack of research on the value of wetland enhancement activities in South Africa, or a gap in the published research on the value of wetland enhancement in South Africa. In a review of the South African literature, including unpublished reports, three specific cases of research into the value of wetland enhancements in South Africa were identified (van Zyl et al., 2004; Pollard et al., 2008; Gull, 2012), all three are reported in the grey literature (two reports and a student thesis).

All the conclusions drawn from the review apply only to trends observed in the published literature. Furthermore, only publications accessible through electronic databases were reviewed, as such older research may have been missed. However, the field of ecological restoration is considered a ‘young’ discipline (Suding, 2011) and the economics of restoration has been described as a new field “that is not yet established as a formal field of enquiry” (Blignaut et al., 2014:2). Similar reviews of the restoration literature have indicated low numbers of publications in the mid-1990s with a rapid growth in publications since then (Wortley et al., 2013; Blignaut et al., 2014). It is unlikely that a significant portion of research has been excluded by considering only publications accessible through electronic databases.

As noted by Pickering and Byrne (2014), bias towards ‘positive’ results in the literature can be a further limitation. In the context of the valuation of wetland enhancement activities, an over-representation of studies where social benefits or positive net benefits of wetland enhancement were found is likely. The present review did not examine value estimates or draw any conclusions about the value of wetland enhancement activities, but rather was a focus on the methods applied, attributes valued and the purpose of the studies.

The selection of search terms is a potential source of bias and study limitation. To try and address this issue, existing literature reviews in the field of ecosystem restoration were examined to inform the selection of search terms. In addition, numerous searches using various search terms and groupings of search terms were undertaken in a ‘trial and error’ approach to determine which terms produced the most relevant results. Using very broad terms resulted in a large quantity of results that included many publications not relevant or meeting the search criteria. For example, a search of Science Direct using the terms “wetland AND (restoration OR rehabilitation OR restore) and (economic OR value OR benefit)” returned close to 1000 publications, many of which discussed the benefits or values of wetlands, without applying any estimation methods. The term ‘valuat*’ was used instead of value, to limit the search to applied valuation studies rather than discussions of 'value'. While an effort was made to include synonyms for the key search criteria (e.g., wetland and marsh; restoration, rehabilitation, creation; and multiple valuation and economic related terms), publications using other terms and descriptions, would have been missed. It is noted that the terms 'constructed', 'mitigation' and 'repair' have also been used in referring to wetland enhancement activities and 'heathland' 'fen meadow' 'peat' and 'peat land' have been used to describe wetland ecosystems. These potential search terms are options for future research or an extension of this literature review.

While cost-effectiveness and economic impact analyses reported in the publications reviewed were recorded and discussed, the focus of this literature was the valuation of benefits, as such the review is not an exhaustive account of cost-effectiveness and economic impact assessments of wetland enhancement activities.

3.5 CONCLUSION

A systematic quantitative review of the English language published literature was performed to examine the extent and characteristics of existing research on the economic analysis of wetland enhancement activities such as wetland restoration, rehabilitation and creation. The existing body of research is relatively small, but growing. Research on the topic remains geographically skewed toward North America and Europe. Africa and South America were notably under-represented; no studies from South Africa were identified. A cross-disciplinary interest in the topic was apparent with economic assessments being performed by both economists and non-economists, both separately and in collaboration. An examination of the methods applied and attributes studied revealed that wetland enhancement valuation studies are relatively heterogeneous across both valuation objects and valuation methods. Greater attention has been given to the habitat provision, water quality improvement and recreation attributes of wetland enhancement, while cultural, educational and disturbance regulation benefits have received little attention. Valuations were most often based on a single attribute of the wetland enhancement. Stated preference methods were most commonly applied and revealed preference methods were least commonly applied. Valuation methods appear to be closely linked to the attributes or benefits under consideration.

A notable finding of the review is the relatively greater attention given to estimating the potential value of wetland enhancement (pre-intervention assessments) relative to quantifying and valuing realized benefits (post-intervention assessments). The result lends support to previous claims in the literature of a gap in the knowledge on evidence-based post-intervention valuation of ecological restoration. The finding has particular relevance in the context of the value transfer method of benefit valuation which involves estimating the value of a wetland enhancement activity by using (transferring) values from another (or multiple other) sites. Without a base of comprehensive post-intervention valuations to draw from, the reliability of the value-transfer approach is questionable.

With regard to research methodology, the publications generally adopted a neoclassical economics approach to evaluate the wetland enhancement activities, except for the few publications which adopted a non-monetary approach (emergy analysis, habitat value assessment and a simple ranking of importance). The choice of search terms is likely to have biased the review towards neoclassical economic studies (e.g., terms such as ‘willingness to pay’, and ‘cost-benefit’); however, the result could point to a prevalence of neoclassical economic methods in the applied valuation of wetland enhancement activities or a lack of attention to applied studies of the value of wetland enhancement by heterodox economists.

The review is limited in several ways and the findings and conclusions should be interpreted accordingly. The search was restricted to English language published studies accessible through electronic databases. As with other literature reviews, the present review was subject to various sources of bias such as the over-representation of studies reporting ‘positive’ results; in this context, studies where social benefits or positive net benefits of wetland enhancement were found. The set of search terms used is a potential source of bias.

Future research is needed to address several knowledge gaps, notably (a) the under-representation of Africa and South America in the published literature on the topic, (b) the lack of research attention given to the post-implementation valuation of the (realized) benefits of wetland enhancement activities, and (c) the seemingly minimal contribution of heterodox economic schools of thought to the applied valuation of wetland enhancement.

Future research on the topic will contribute to growing the relatively small base of literature on the economic analysis of wetland enhancement, thereby providing evidence and data to support the development of meta-analysis and value transfer approaches, which may be useful practical tools for decision-making. Given the interdisciplinary nature of the 'economics of wetland enhancement' collaborations between disciplines are encouraged in future research on the topic.

The following three chapters present the chosen cases studies. The context, methods, analysis and results are described for each case evaluation. The final chapter (Chapter 7) synthesises the research findings and presents the framework for the valuation of wetland rehabilitation. The first case study demonstrates a post-rehabiliation evaluation that illustrates the economic valuation of the contribution of rehabilitation to wetland-based livelihood activities within a communal land context.

CHAPTER 4: MANALANA WETLAND REHABILITATION: A COST-BENEFIT ANALYSIS OF THE CONTRIBUTION TO LOCAL LIVELIHOODS

4.1 INTRODUCTION

Wetlands remain an important part of the livelihood[52] strategy of many marginalised people who live near them (Kangalawe and Liwenga, 2005; Rebelo et al., 2010; Kumar et al., 2011; Black et al., 2016). For communities located in rural areas, the link between people's livelihoods and natural ecosystems is often particularly direct with households dependent on natural ecosystems to provide resources for meeting their basic needs (Rebelo et al., 2010; Musyoki, 2012). In this way, wetlands can be the setting from which people derive their livelihoods and enable them to cope in times of stress and to assist others (Kumar et al., 2011; McCartney et al., 2011; Mitsch and Gosselink, 2015). This is the case in South Africa, where wetlands contribute directly to people's livelihoods by providing life-support functions including the provision of food, water, medicines, fibre and fertile lands for cultivation (Pollard et al., 2005; Lannas and Turpie, 2009; Adekola et al., 2012; Black et al., 2016).

Consequently, the degradation of wetland systems has direct negative impacts on the livelihoods of wetland dependent households. Wetland degradation jeopardizes the ecological integrity of these systems reducing their capacity to provide the ecosystem functions and services that support livelihood benefits (Schuyt, 2005; Pollard et al., 2005; Kumar et al., 2011; Black et al., 2016). Wetland rehabilitation, on the other hand, can improve the ecological integrity of a wetland, or halt further degradation of the system, thereby enhancing or securing the livelihood support benefits of wetlands for rural communities and marginalised groups, wetland management and poverty can be inextricably linked.

This is the case for the Manalana wetland which plays an important role in the livelihood strategy of many of the community members, through the provision of material resources, in an area with high levels of poverty (Pollard et al., 2005; Pollard et al., 2008). The size and integrity of the Manalana wetland has been severely impacted by erosion. Assessments

---

[52] 'Livelihood' is defined by Hay et al. (2014:8) as "the means of securing what we need to live – food, money, shelter, safety, education, connectedness".

undertaken more than 10 years ago called attention to the livelihood benefits of the wetland and how these benefits were at risk from continued erosion of the wetland. At the time, farmers of a wetland downstream of the Manalana wetland lost their fields due to the advance of an erosion gulley by nearly 1000 m in a single year (Pollard et al., 2008). In the Manalana wetland itself, several portions of the wetland had been lost through erosion; however the remaining intact portions continued to support livelihood activities. It was against this backdrop, that a decision was taken to rehabilitate the Manalana wetland with the primary intention of halting the erosion proliferating through the wetland to secure the remaining areas of functional wetland habitat and safeguard the associated livelihood benefits (Bothma, 2004). A pivotal driver of the decision to rehabilitate the wetland was the expectation that in the absence of the rehabilitation, severe erosional degradation would proceed through most of the remaining intact wetland reducing its capacity to provide the ecosystem functions and services that support local livelihoods.

As part of an evaluation undertaken eight years after the rehabilitation was completed, this case study addressed the economic valuation of the contribution of the rehabilitation to three primary wetland-based livelihood activities: crop cultivation, livestock grazing and reed harvesting[53]. The purpose of the evaluation was to establish whether the objectives of the rehabilitation had been achieved, and the aim of the economic valuation component of the evaluation was to determine whether the rehabilitation investment generated an economic efficiency gain based on achieving the primary objective of the rehabilitation. The intention of the study was to derive an economic value estimate of the specific contribution of the rehabilitation to local livelihood activities, measured through a monetary metric, and compare the value to the cost of the rehabilitation.

There are relatively few published studies of the economic value of the specific contribution of rehabilitation to the provision of material resources associated with local livelihood

[53] This case study was part of the Water Research Commission Project K5/2344 and preliminary parts of the analysis were reported in deliverable 9 (2018) "An integrated report detailing the monitoring and evaluation of the rehabilitation undertaken at the Greater Edendale Mall and Manalana wetland sites". The evaluation was undertaken by D Walters and M Browne with contributions from D Kotze, C Cowden, M Grewcock, B.Makhabela.

activities (Chapter 3)[54]. Contrary to this trend, 82% of the studies identified in a meta-analysis of the economic value of wetlands in *developing countries*[55] valued the provision of material resources (Chaikumbung et al., 2016). The majority of these studies estimated direct use values of material resources derived using a market-price approach.

Several studies have assessed the economic value of the contribution of wetland-based activities to local livelihoods in southern and eastern Africa including crop cultivation, livestock grazing and watering, and the harvesting of wetland vegetation for household materials, cultural uses and traditional medicines (e.g., Emerton et al., 1998; Turpie et al., 1999; Mwakaje, 2009; Nabahungu and Visser, 2011). In South Africa specifically, Lannas and Turpie (2009) and Adekola et al. (2012) analysed the economic value of livelihood benefits derived from the provisioning services of wetlands in an urban informal settlement (Western Cape) and rural (Limpopo) context, respectively[56]. Both studies applied market-based valuation techniques using data on household wetland use, socio-economic information and market prices for wetland-related products collected through a combination of participatory approaches, field observations and household surveys.

Lannas and Turpie (2009) found that the wetland performed a safety-net function for a small proportion of the adjacent community, mainly through providing livestock grazing in a largely transformed landscape. Adekola et al. (2012) determined that the wetland contributed significantly to household subsistence and as a source of income through a range of material resources, with cropping accounting for the largest value contribution and sedge collection as the primary source of cash income. Hay et al. (2014), however, suggested that the reliance on wetlands for consumptive use in South Africa is declining, but emphasized that where this form of use remains, it is often part of a vital livelihood strategy.

The studies of Lannas and Turpie (2009) and Adekola et al. (2012) considered the economic value of the total livelihood benefit of the wetland as an indication of its importance to the

[54] However, published studies do not necessarily reflect to what extent the provision of material resources to local communities is considered in wetland rehabilitation decision-making.
[55] Of the total set of studies identified, 21% were from Africa (excluding North Africa).
[56] The assessment of the urban wetland by Lannas and Turpie (2009) was part of a comparative study of a rural wetland in Lesotho and the peri-urban wetland in the Western Cape.

surrounding communities and the potential implications of a complete loss of the wetland resources. The economic valuation of wetland rehabilitation, however, requires a marginal valuation approach which necessitates a comparison of the difference in benefits between the 'with' and 'without' rehabilitation states informed by an understanding of the biophysical processes of the wetland. In this case study, this was achieved through biophysical and social assessments conducted eight years after the rehabilitation was completed. The 'without rehabilitation' scenario was conceptualized, in consultation with ecologists familiar with the site, drawing on evidence from studies of the Manalana wetland particularly the PhD research outputs of Ngetar (2011) who investigated the causes of wetland erosion at Craigieburn and Riddell (2011) who analysed the hydrological processes of the Manalana wetland, and observations of the extent of wetland habitat loss in other parts of the catchment.

The chapter is structured as follows. The study area and wetland scenarios are presented followed by a description of the data collection and valuation methods. The results of the valuation are reported and compared to the costs of the rehabilitation in a partial CBA. The sensitivity of the cost-benefit outcomes to several factors are explored. A detailed discussion of the findings follows, with attention to contextualizing the results within local household incomes and the findings of similar studies, and highlighting emerging questions and limitations. Conclusions are presented in the final section.

4.2 STUDY AREA AND WETLAND SCENARIOS

The Manalana wetland is situated within the Manalana River catchment, an area of roughly 1.5 km^2 in the eastern foothills of the Northern Escarpment in Mpumalanga Province, South Africa. The Manalana River is a minor tributary of the Sand River. The Manalana catchment comprises the peri-urban residential settlement of Craigieburn, dry-land and wetland cropping areas, a network of roads and pathways, and natural vegetation. The distribution of natural vegetation and human development is near equal across the catchment (Walters and Browne, 2018). Land tenure in the Craigieburn area is communal based on traditional community arrangements (Pollard et al., 2005; Ngetar, 2011). Forced removals of the African population, because of the political situation at the time, resulted in a significant increase in the population of Craigieburn between 1965 and 1974 (Pollard et al., 2005). After the abolishment of the apartheid system (1994) such areas became communal lands.

Craigieburn Village is a suburb of Acornhoek and falls within the Bushbuckridge Municipal Area. Socio-economic statistics for Bushbuckridge suggest that households remain vulnerable: employment rates are low, poverty rates are high, household incomes are lower than the provincial average and social grants are the primary source of income for many households (Bushbuckridge Municipality, 2014). Service delivery is characterised by extensive backlogs and the natural environment of the area is recognized as playing an important role in providing services to residents (Bushbuckridge Municipality, 2014). This is the case for many of the residents of Craigieburn who rely on the Manalana wetland for natural resources and favourable cropping conditions.

A household survey in 2004, found that 70% of residents within the greater area used wetland-related resources to meet their livelihood needs (Pollard et al., 2005). It was observed that the intact or functional portions of the Manalana wetland were used by local residents in a number of livelihood activities including crop cultivation, livestock grazing and reed harvesting. The overriding profile of wetland users was that of women between 35 and 70 years of age - mainly from single-headed households. At the time, local livelihoods were considered to be vulnerable to economic shocks and stresses such as the loss of a household member and drought.

4.2.1 The ‘without rehabilitation’ scenario

Prior to rehabilitation, ecological surveys had identified two gullies within the Manalana wetland which were actively eroding and continuing to advance upstream into intact portions of the wetland (Bothma, 2004). These erosion gullies create a ‘soil gradient’ between the upstream intact portions of wetland and the downstream degraded areas and effectively soils flow out of the wetland into the downstream system. This results in a reduction in wetland habitat and viable substrate for cultivation. In addition, hydrological monitoring of the system showed that the gullying of the wetland had created a desiccating environment adjacent to the upstream end of the gullies (Riddell et al., 2012). The overall effect of the gully erosion had been a reduction in wetland habitat and reduced soil moisture in parts of the wetland. A decline in fertility due to the loss of organic matter and nutrients through soil loss is also associated with gully erosion (Pollard et al., 2005).

A comprehensive study of the causes of erosion within and along the Craigieburn wetland attributed the initiation of the erosion to prevailing geomorphic processes (Ngetar, 2011). The research showed that the erosion within the wetland is part of a long-term process that predates human impacts and is likely driven by long term climatic variables that influence catchment sediment and water delivery responses. According to Ngetar (2011), the erosion process within the system is enhanced by current warm and moister climate patterns resulting in higher rainfall, occasional catastrophic rainfall events and increased stream erosivity. Measurements of aerial photographs from 1954 to 1997 indicated that the upper gully migrated headward at an average rate of 0.75 metres per year, while the lower gully eroded at an average rate of 13 metres per year. More recent measurements suggested a rate of 36 metres per year for the upper gulley and 23 metres per year for the lower gulley, with much of the erosion occurring in the high rainfall months (Ngetar, 2011). Erosion of the Manalana wetland system over time has greatly reduced its size and spatial extent. Soil samples indicated that the wetland system once covered a much larger area; average measurements at the upper portion of the wetland, for example, showed that it had shrunk laterally in size, 15 metres from the then current extent (2005/2006) (Ngetar, 2011).

The research of Ngetar (2011) indicates that the Manalana wetland system is naturally dynamic, undergoing erosion and depositional processes over time. The physical attributes of the system, particularly the geomorphological setting, suggest that the wetland is inherently vulnerable to erosion (Walters and Browne, 2018). The two primary erosion gullies in the wetland are 'active' and, prior to the rehabilitation intervention, were spreading upstream and laterally through the wetland system (Ngetar, 2011).

Based on the evidence from the field studies of the system, particularly those of Ngetar (2011), Riddell (2011) and Walters and Browne (2018), it is highly probable that in the absence of the rehabilitation structures, severe erosional degradation would have proceeded through most of the remaining intact portions of wetland resulting in a significant reduction in the extent of functional wetland habitat. Consequently, the area (viable substrate) available for wetland cropping and the extent of natural vegetation available to support reed harvesting and livestock grazing would have been greatly reduced. Observations during the various surveys of the wetland found very limited cultivated patches remaining in eroded portions of the wetland, indicating that once an area becomes eroded, it is no longer cultivated.

Given the findings of the comprehensive research related to the study site and observations of the extent of wetland habitat loss in other parts of the catchment[57], the following assumptions of the 'without rehabilitation' scenario were made:

- A 75% reduction in available area for crop cultivation;
- A 75% reduction in availability of wetland reeds; and
- A 75% reduction in the area of wetland vegetation available for livestock grazing.

4.2.2 The 'with-rehabilitation' situation

The intention of the rehabilitation intervention was to secure the integrity of the remaining intact portions of the Manalana wetland, rather than to reinstate integrity to the degraded portions. The wetland rehabilitation efforts included the installation of two erosion control structures in the system during the latter half of 2006: an impervious concrete weir to address a lesser gulley head threatening an upstream portion of the wetland and a pervious gabion weir to address a greater gulley head threatening a lower portion of the wetland (Portion 1 and 2 respectively in Figure 4.1) (Bothma, 2004).

During the post-rehabilitation evaluation, the structures were found to be intact with no indication of damage or structural failure and appeared to have been functioning as anticipated in the rehabilitation plan (Walters and Browne, 2018). The structures had deactivated the targeted erosion that existed prior to rehabilitation and no further erosion or movement of the gullies was evident. In this case, the rehabilitation structures have 'suspended' the erosion processes by introducing new artificial base levels, preventing the gullies from propagating into the intact wetland portions above the structures (Walters and Browne, 2018). According to Riddell et al. (2012), the groundwater hydrology of the wetland

[57]Ecological assessments were undertaken across the Blyde and Sand River catchments during a survey by the Rennies Wetland Project with the National Working for Wetlands Programme, AWARD and the Mpumalanga Department of Economic Development, Environment and Tourism, (1998–2000).

is controlled by the distribution of clays within the wetland; loss of these clays, through erosion for example, negatively impacts on the system's hydrology. In the case of the concrete weir, Riddell et al. (2012) showed that the rehabilitation structure had restored the hydrological condition of the system by 'mimicking' a clay plug at the base of portion 1 which had been eroded. In the case of the gabion, the structure had protected the wetland's hydrology by preventing the loss of the clay plug at the lower end of portion 2.

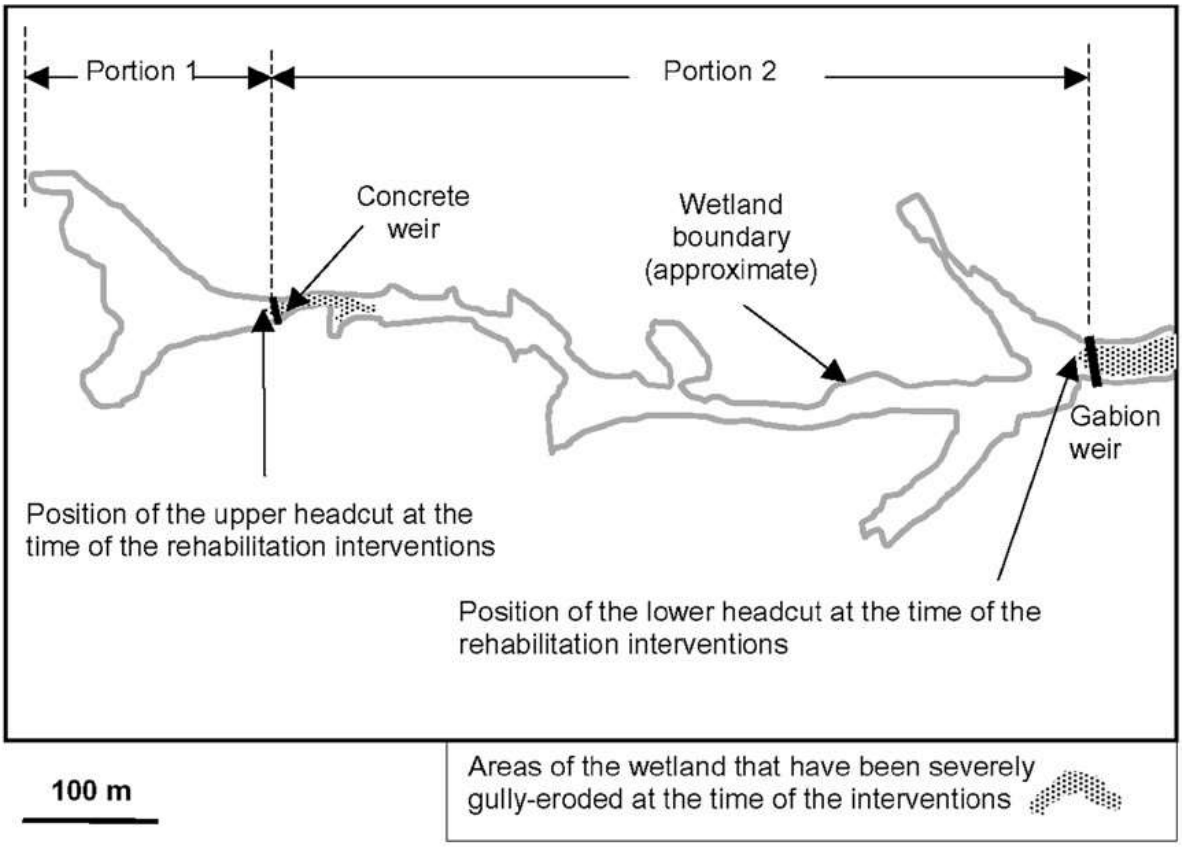


Figure 4.1: Schematic map of the Manalana wetland showing the location of the two erosion gulleys and the rehabiliation structures.
Source: Reproduced from Kotze et al. (2008:52).

A summary of the objective and evaluation result for each of the erosion control structures is provided in Table 4.1. The findings corroborated earlier observations that no additional gully erosion had occurred upstream of the structures (Riddell et al., 2012). However, there was no indication at either structure that their function (i.e., erosion control) could, in the foreseeable future, be replaced by natural processes, suggesting that the integrity of the remaining wetland habitat is inextricably linked to the presence and functioning of the structures (Walters and Browne, 2018).

Table 4.1: Summary of the objectives and evaluation result of the Manalana wetland rehabilitation structures

Problem	**Intervention**	**Objective**	**Evaluation (2016)**
Gully head erosion 1-2m deep and 5m wide	Concrete boxed drop inlet weir structure (X32A063)	The structure will stop the gully head erosion from eroding laterally and vertically. The structure will prevent sediment from entering the wetland below the structure.	Intact, performing as planned 
Gully head erosion varying between 1m and 3m deep and 40m wide	Gabion structure (X32A065)  *Source: Kotze et al. (2008)*	The structure will stop the gully head erosion from eroding laterally and vertically. Sediment from the gully will be caught behind the structure.	Intact, performing as planned 

Source: Summarised from the rehabilitation plan (Bothma, 2004) and ecological assessment findings during the in-field evaluation (Walters and Browne, 2018); photos by author unless otherwise stated.

It was concluded that the rehabilitation of the Manalana wetland had secured the provisioning service potential of the Manalana wetland. The rehabilitation structures halted active erosion that threatened two portions of intact (unchanneled) wetland thereby preventing further loss of wetland vegetation and viable substrate for crop cultivation and positively influencing the wetland's hydrological condition. During the post-rehabilitation assessment, one of the wetland farmers described the result of the rehabilitation as having "Helped her a lot. There was a big donga [erosion gulley] just below the field, which was getting closer. The gabion stopped the donga, the field is wetter now which is good for the madumbe".

4.2.3 Comparison of the two wetland cases

A description and comparison of the two cases is summarized in Figure 4.2. The primary difference between the two states, underpinning this economic valuation study, is the difference in extent of intact wetland area available to support cropping (e.g., viable substrate), reed harvesting and livestock grazing. No assumptions were made of the likely impact of continued erosion on the yield of cultivated crops and wetland resources for any remaining wetland portions (without rehabilitation) as there have been no measurements of yields under such conditions.

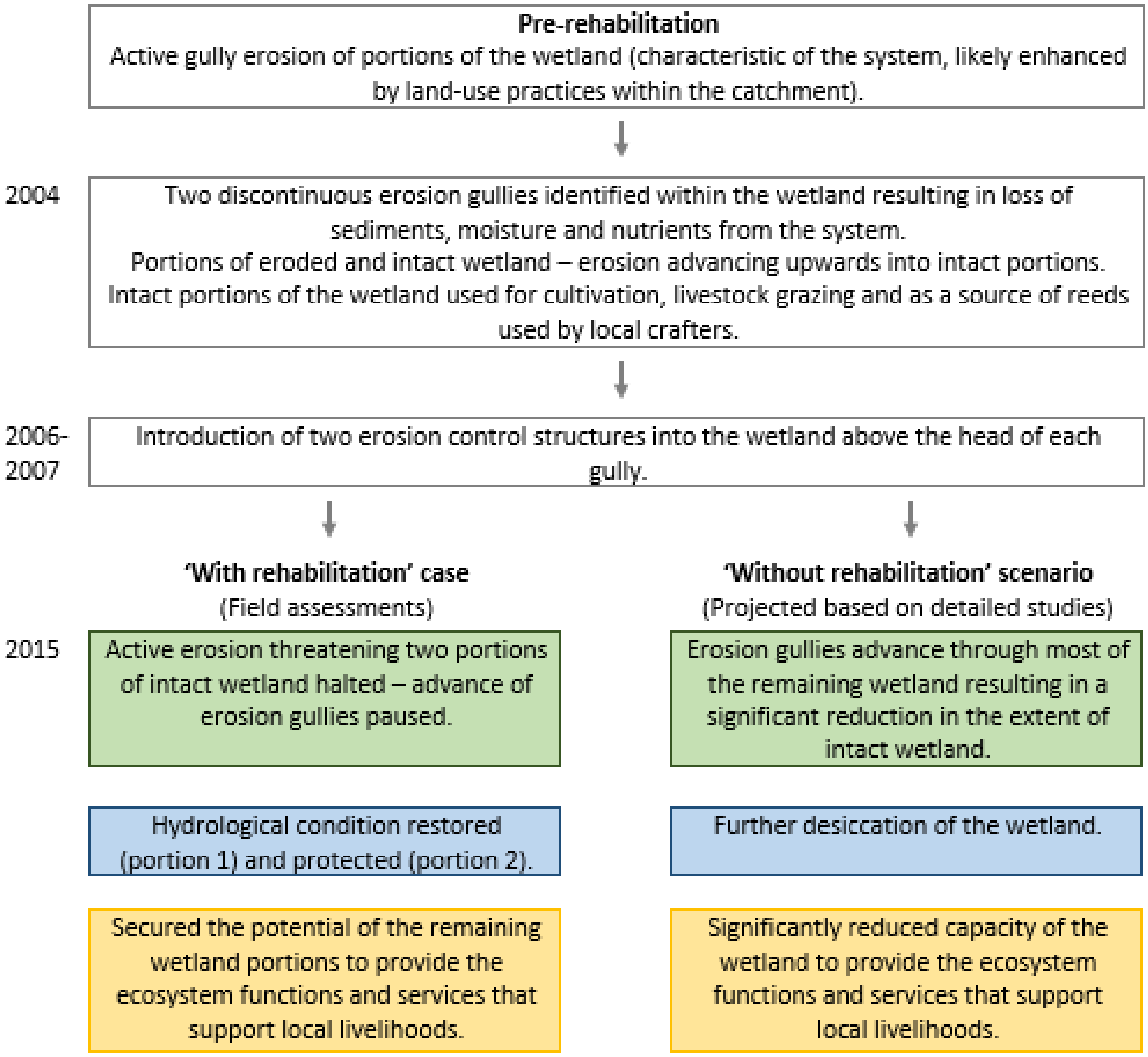


Figure 4.2: Comparison of the 'with' and 'without' rehabilitation cases for the Manalana wetland.

4.3 METHODS AND DATA COLLECTION

4.3.1 Overview

The focus of the case study was the contribution of the rehabilitation to three primary wetland-based livelihood activities at the village level: crop cultivation, livestock grazing and reed harvesting. This focus aligns with the specific goal of the rehabilitation intervention to secure the livelihood benefits of the wetland. Following the TEV typology, these activities would be classed as having 'direct (consumptive) use value'. Direct consumptive use values, measured through a monetary metric, were estimated for these three activities. A marginal value framing was adopted, which is appropriate for evaluating the specific contribution and value of the rehabilitation, whereby the difference in benefit levels between the 'with' and 'without' rehabilitation cases were estimated.

In this case, the aim of the rehabilitation intervention was to prevent a decline in the potential of the wetland to support local livelihood activities. A reference, or baseline, level of benefits was established based on information available from pre-rehabilitation studies of the wetland and household surveys. In the post-rehabilitation case, it was expected that the capacity of the wetland to provide this level of benefits would be maintained; under conditions of continued degradation, it was projected that the capacity of the wetland to provide this level of benefits would be greatly reduced. The post-rehabilitation current use of the wetland was investigated through primary data collection undertaken as part of this case study. The supply, demand and monetary value of each of the three primary activities (crop cultivation, livestock grazing and reed harvesting) was assessed. Based on the evidence presented in Section 4.2.1, the reduction in benefits under the degraded scenario was projected as a 75% decline in the extent of substrate suitable for cultivation and a 75% decline in natural vegetation available for reed harvesting and livestock grazing relative to the pre-rehabilitation baseline.

Using both secondary and primary data, direct use values of wetland-related resources associated with local livelihood activities were estimated for both the 'with' and 'without' rehabilitation cases. The difference in values was taken as the benefit of the rehabilitation and compared to the costs of the rehabilitation in a partial CBA. The cost of the rehabilitation was based on information provided by the project implementers from the rehabilitation planning and budgeting processes. In the specific case of the Manalana wetland, a 25-year period is considered the realistic life of the rehabilitation structures considering that the structures do not receive regular maintenance (Walters and Browne, 2018). Net present values were estimated under a 50-year timeframe to consider the sensitivity of the results to the selected timeframe. As the choice of the appropriate discount rate is a subject of debate, three different discount rates were applied. Frequently, the usual discount rate for public investments for the relevant country is applied in ecosystem valuations. In South Africa, an 8% discount rate was considered appropriate at the time of the assessment, following Mullins et al. (2014). However, from a sustainability perspective, an argument exists for discounting the future benefits that flow from ecosystems at a lower rate, or not at all (Fenichel et al., 2017). As such, a 3% and 6% discount rate were applied as a sensitivity analysis. The sensitivity of the results to several other factors were also explored and the details are reported with the results (Section 4.4.4).

4.3.2 Data collection

Information was derived from both secondary and primary sources. Prior to the wetland rehabilitation, several ecological and social assessments of the study area had been undertaken and the results were used to establish the reference level of wetland benefits. Secondary data sources included: results of surveys and data collection conducted between 2003 and 2007, as discussed in Pollard et al. (2005) and Pollard et al. (2008) which included information on wetland uses, the characteristics of wetland users, and cropping, reed harvesting and grazing practices and production quantities; the wetland rehabilitation plan (Bothma, 2004); findings of ecological assessments described in Kotze et al. (2008); and the research outcomes of Ngetar (2011) and Riddell (2011) which particularly provided the evidence for defining the ‘without rehabilitation’ scenario (Section 4.2.1). The study approach and data sources are outlined in Figure 4.3.

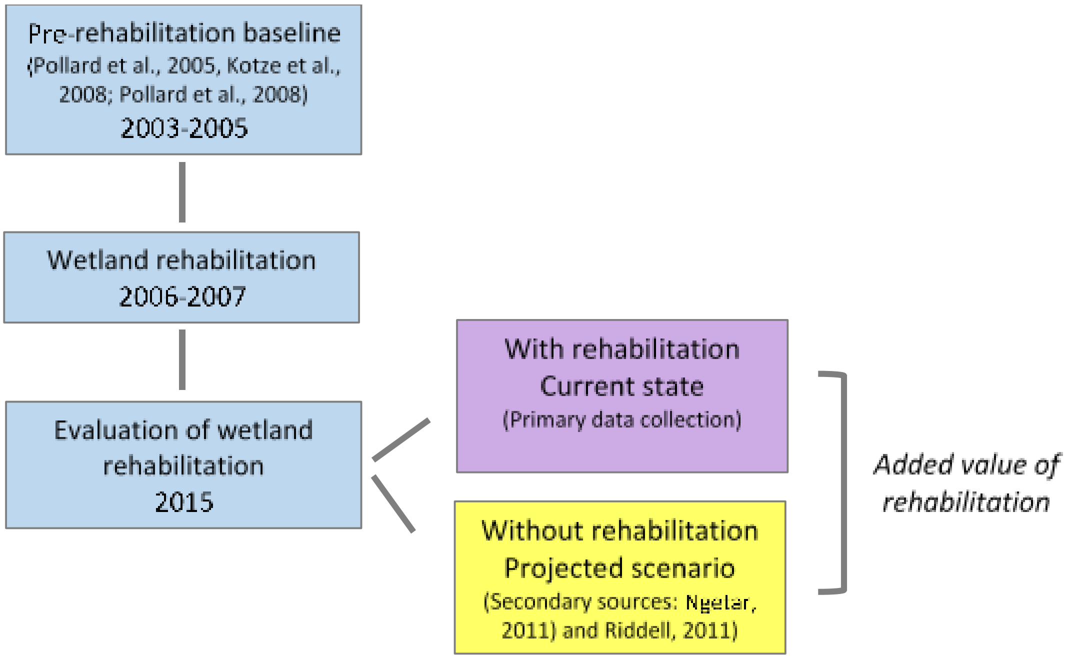


Figure 4.3: Approach and data sources for evaluating the added value of the rehabilitation of the Manalana wetland.

Primary data collection was conducted during a four-day site visit as part of a post-rehabilitation evaluation undertaken eight years after the rehabilitation structures were completed. Data was collected through focus group sessions, semi-structured interviews, observation and narrative description through landscape walks with wetland users, and in-field measurements including the mapping of crop fields and wetland attributes and plot level measurements. Prior to the site visit, permission to conduct the research was sought through the appropriate community processes to legitimise the study ethically. These introductions and the community engagements were facilitated by a local resident who had been part of the team that conducted the pre-rehabilitation surveys.

Drawing from existing information and the findings of the pre-rehabilitation studies, a set of themes and questions were developed to guide the primary data collection. Figure 4.4 provides an overview of the key themes and activities. The primary data collection activities aimed to:

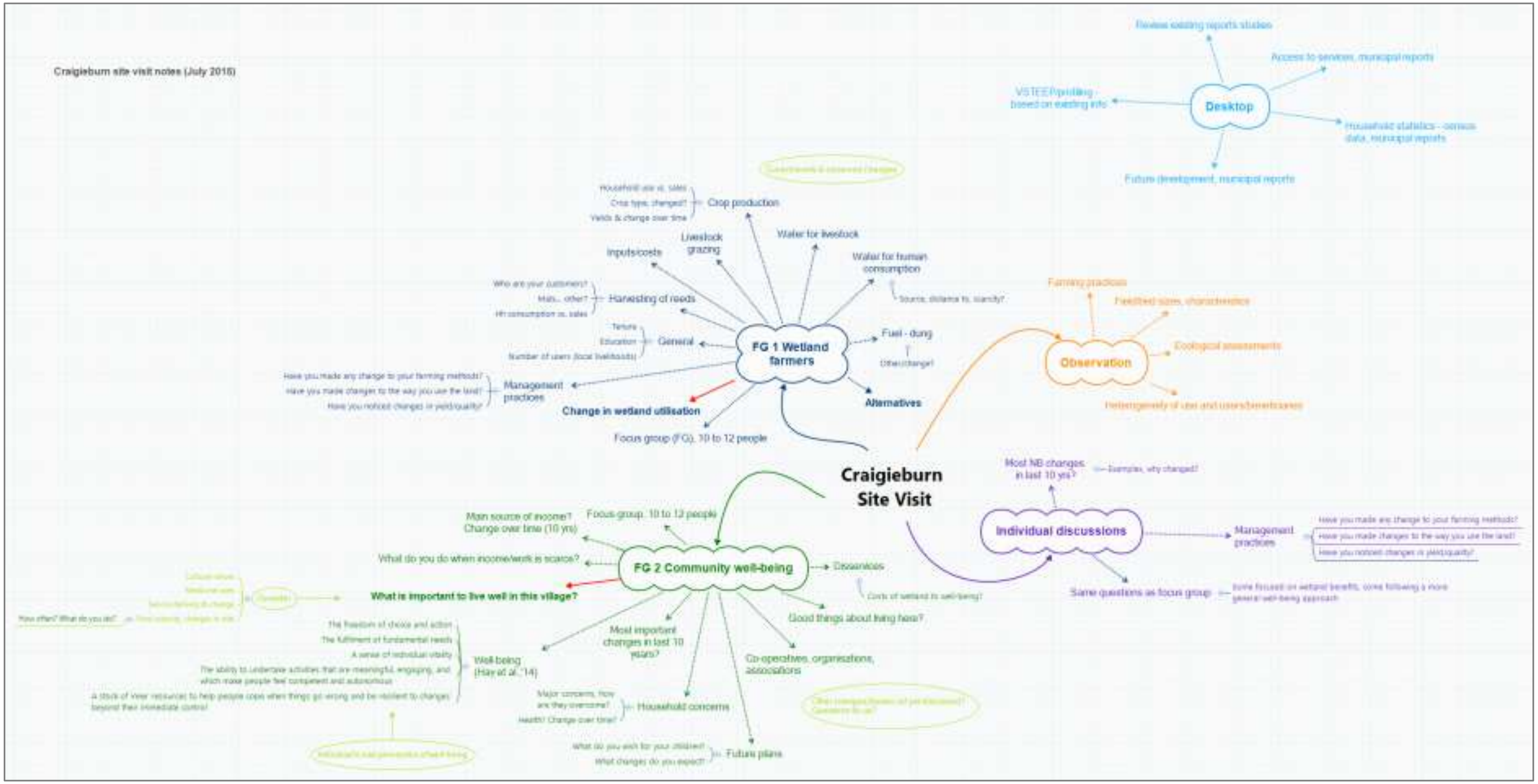
- Establish the use and livelihood-related benefits of the wetland for the local community and explore how use may have changed since the pre-rehabilitation baseline;

- Identify the demographic characteristics of wetland users and the motivators or drivers of wetland use and consider ‘access’ to wetland use/benefits (e.g., tenure and governance elements) and alternative livelihood options;
- Explore the perspectives of the community on the wetland and the rehabilitation and the connections to their well-being;
- Collect data on wetland crop production, reed harvesting and livestock grazing, prices and inputs/costs and specific information on any additional benefits identified.

Two focus group sessions were held (Table 4.2), each with a different participant base (all participants were local residents). Individual interviews were conducted with four wetland users and focussed on obtaining details on crop production, reed harvesting and livestock grazing and trends in wetland use. Field observations and measurements were undertaken and GPS coordinates and details of the field and plot sizes were recorded. Photographs from the field work are presented in Figure 4.5.

Table 4.2: Details of focus group sessions, Craigieburn Village, July 2015

	Focus group 1	**Focus group 2**
Date	21/07/2015	22/07/2015
Duration	5 hours	4 ½ hours
Approach	Semi-structured	Semi-structured
Number of participants	15	17
Description of participants	Wetland farmers (14 female, 1 male)	Mixed group of community members (across gender, age and livelihood)



Note: FG is focus group.

Figure 4.4: Summary of key themes and questions guiding the primary data collection activities.

Participants contributing during Focus Group 1, Craigieburn Village, July 2015.

A crate which is used to sell madumbes, Craigieburn Village, July 2015.

Wetland fields identified along the Craigieburn system were investigated in terms of their plot and bed size and the current level of use, Craigieburn Village, July 2015.

Figure 4.5: Photographs from the site visit, Craigieburn Village, July 2015.

4.3.3 Economic valuation methods

The market price valuation method was applied in the case of wetland crop production and reed harvesting. This is a common approach employed to estimate direct use values where goods are traded in a market (Turner et al., 2008). Application of the method entailed multiplying the quantities of crop produced and reeds harvested by the local unit prices for each commodity to generate an annual gross margin value. Local market prices and production data were obtained through the primary data collection activities and informed by secondary sources. In the case of crop production, estimated total yield was reduced by 20% to account for retention (for replanting) and wastage.

The monetary value of reed harvesting was estimated based on the price of a bundle of reeds and the quantity of bundles harvested annually. Wetland users harvest reeds from their fields, which include patches of reeds and crop beds, and reported that on average, $1m^2$ of wetland reed area produced one bundle of reeds. Based on the detailed surveys undertaken prior to the rehabilitation (Pollard et al., 2008), it was assumed that reeds are harvested from 70% of the wetland fields; that 70% of the reed area per field is harvested each year; and that 20% of the harvest is wastage.

Wetland fodder is not traded and has no market price. One valuation option is to apply the price of grass hay, which is traded in the market, as a replacement cost approach. In the context of the Craigieburn community however, it is unlikely that livestock owners would be willing or able to pay for hay to replace the 'free' fodder of the wetland. Instead, the condition of livestock would decline either through lack of fodder or through having to range further distances to reach alternative grazing options.

In South Africa, subsistence grazing practices are seasonal, driven by fodder production in response to rainfall (Shackleton et al., 2008). Often, livestock depend on key resource areas such as wetlands and agricultural residues in fields during the dry season. This is the case for the Craigieburn livestock owners who use the Manalana wetland for grazing during the dry season at a time when the availability and quality of fodder in the surrounding grasslands is low. Without access to resources such as wetlands, livestock lose condition and body mass during the dry season and mortality increases (Shackleton et al., 2008). In the case of the Manalana wetland, wetland grazing is a coping strategy during a time when few alternatives are available and arguably the value of the wetland is greater than the amount of fodder grazed. It is the timing of the supply of fodder that is particularly critical and the wetland plays an important role in maintaining stock condition and reducing mortality through the dry period.

To circumvent the challenges of assessing changes in livestock condition as a direct result of access to wetland fodder (or less thereof), Pollard et al. (2008) proposed that the value of wetland grazing could be reflected by the number of livestock that the wetland is able to sustain in a healthy condition over a critical period when the availability of grassland fodder is low. This approach was employed in the present study to estimate the value of the rehabilitation for livestock production following the technique and assumptions outlined by Pollard et al. (2008).

Owing to favourable moisture conditions, forage quantities in wetlands tend to be higher than that of adjacent grasslands. A study of wetlands in the Sani Pass region of the Drakensberg, for example, found fodder production of the wetlands to be as much as double that of the non-wetland areas (Morris et al., 1989, cited by Pollard et al., 2008). However, this is not always the case and depends on the specific conditions of the wetland including the level of wetness and presence of palatable species and the quality of fodder of the non-wetland areas. In the case of the Manalana catchment, conditions are such that Pollard et al. (2008) suggested that the wetland could provide 75% more forage, on a unit area basis, relative to the surrounding grasslands. Cattle, large stock units (LSU) are the predominant livestock grazing the Manalana wetland. The following assumptions were made in calculating the number of cattle that the wetland could sustain through the critical dry period:

- Fodder production for the surrounding grassland areas, based on the mixed veld classification of the area, of 1 800 kg/ha/year;
- The Manalana wetland produces 75% more forage than the grassland areas (3 150 kg/ha/year);
- Fodder production was reduced by 20% to account for the relatively poor condition of the Manalana wetland vegetation (2 520 kg/ha/year);
- 30% of the available fodder is grazed before the critical dry period;
- 30% of the available fodder is not grazed (during the critical period);
- A fodder consumption of 10 kg dry matter per LSU/day; and
- A 30-day critical period of dependence on the wetland for grazing.

In order to give value to the number of cattle that could be sustained by the wetland, a value-transfer approach was taken and the annual direct use value per animal[58] (R/LSU/year), estimated in a separate study of the Sand River Catchment (Shackleton et al., 2005) and adjusted for inflation, was applied. The value of the rehabilitation was taken as the value of

[58] The annual direct use value per animal was derived in terms of a range of goods and services associated with cattle ownership including herd growth (savings), milk, manure, slaughtering, cash sales and culture-related benefits (Shackleton et al., 2005).

the difference in the number of livestock that could be sustained through the dry period between the ‘with’ and ‘without’ rehabilitation states.

4.4 RESULTS

The intact (non-eroded) portions of the Manalana wetland are a matrix of cultivated fields and natural vegetation; land cover is classified as subsistence agriculture (45%), natural vegetation (54%) and a road (0.5%). The surrounding area of the catchment consists of natural vegetation (54%), peri-urban development (37.5%) and cultivated fields (8.5%) (Walters and Browne, 2018). The slopes surrounding the Manalana wetland contain grazing land and agricultural fields, with homes accompanied by household gardens situated on the flatter areas and ridges. Residences are located within 400 meters of the wetland and the wetland is easily accessed by both people and livestock.

Crop cultivation, livestock grazing and reed (*Schoenoplectus brachyceras)* harvesting are the main livelihood activities associated with the Manalana wetland. The site investigations, focus group discussions and semi-structured interviews confirmed that the wetland is used primarily for cropping purposes, with some households harvesting reeds. Livestock (cattle) are grazed in the wetland during the dry season. While the wetland is used for the same purposes as prior to the rehabilitation, this post-rehabilitation study revealed that there have been some changes in the patterns of use and a significant reduction in the number of cattle in the area. The focus group sessions and interviews confirmed that the predominant group of wetland users remains women, many of who are older than 70 years. There appear to have been few new wetland farmers over the last 10 years, and the number of farmers has declined from 34 to 28.

4.4.1 Crop cultivation

Food crops are grown within three distinct areas by Craigieburn households. Wetland areas are primary used for growing madumbe (*Colocasia esculentaI*, internationally known as taro) which is grown in the summer (wet) season. Pumpkins are typically grown with the madumbe on the edges of the wetland beds. Other crops, such as leafy vegetables, are planted by several farmers following the madumbe harvest. The wetland fields provide favourable conditions for cultivation of the moisture loving plant madumbe crop and enable farmers to grow additional crops for a longer period into the dry season compared to dryland fields. While all wetland farmers plant madumbe, not all the farmers plant additional crops.

Dryland areas, typically the slopes above the wetland, are used to grow maize. Household gardens are used for vegetables such as spinach, carrots and tomatoes and for fruit trees. Resident farmers indicated that crops are needed in different places to ensure both sufficient food quantity and variety. Nabahungu and Visser (2011) similarly observed a linkage between farming in wetlands and drylands in their study of a wetland in Rwanda.

Farmers cultivate the wetland primarily to satisfy household food requirements, with excess produce sold to generate income. Produce is largely sold to other residents, but some farmers also sell excess, specifically madumbe, in the nearest town. Focus group discussions revealed that madumbe is a preferred crop for local residents; households who do not farm purchase madumbe and other crops from those who do. Participants indicated that all crops are available within the village and there is no need to travel to the nearest town to purchase vegetable produce, particularly as local sellers are cheaper and transport costs to town are avoided by purchasing locally. In this way, all residents depended on the produce from the wetland, not just the farming households.

In this case study, the focus of the monetary valuation of crop cultivation was of the annual value of madumbe production as there is a specific reliance on the wetland for this crop which is moisture loving and not grown in the dryland or household areas. The madumbe yield per unit area was calculated from the information gathered through the interviews with farmers, the focus groups and field measurements. The average yield of 2.5 kg/m^2 per unit area was the same as that estimated for the pre-rehabilitation assessment, suggesting that there had been no significant decline in the production potential since the pre-rehabilitation assessment. The yield estimate is consistent with estimates from a study of wetland madumbe cultivation in KwaZulu-Natal of 2.5 to 3.5 kg/m^2 (Kotze et al., 2002). The value calculation and results are reported in Table 4.3. The difference between the value of production under the ‘with’ and ‘without’ rehabilitation cases, R64 974 per year, is the contribution of the rehabilitation to the direct use value of the wetland for madumbe crop production.

Table 4.3: Contribution of the rehabilitation to the annual direct-use value of the Manalana wetland for madumbe crop production, 2015

Measure	With rehabilitation (2015)	Without rehabilitation (projected)	Data source
Madumbe yield (kg/m^2)	2.5	2.5	Field survey
Crop area (m^2)	6 188	1 547	Field study/projected
Total yield (kg)	15 470	3 868	Calculation
Retention & wastage (%)	20	20	Assumption
Yield after retention (kg)	12 376	3 094	Calculation
Madumbe price (R/kg)	7.00	7.00	Field survey
Value of production (R/year)	86 632	21 658	Calculation
Rehabilitation contribution (R/year)		**64 974**	

Note: All monetary values are 2015 Rands; discrepancies in sub-totals are due to rounding.

4.4.2 Reed harvesting

Reeds harvested from the wetland are sold directly to other households, in bundles, or are woven into mats for household use and for trade. Wetland users indicated that the sale of reed mats was generally to people from outside of Craigieburn Village, often for use in cultural initiation activities. Those wetland users who sell reed mats explained that selling mats is an important source of income; the activity is generally undertaken by women with few alternative sources of income. Discussions during the interviews with farmers and focus group sessions revealed that the wetland continues to be used for reed harvesting.

Reed harvesters confirmed that they had not observed a change in reed availability and quality over the previous eight years. From the interviews with farmers, focus groups, field measurements and observations, and informed by the ecological assessments of the wetland extent and condition, it was concluded that there had been no significant change in the area and yield of reeds relative to the pre-rehabilitation survey. The value calculation and results are reported in Table 4.4.

The difference between the value of production under the 'with' and 'without' rehabilitation cases, R11 995 per year, is the contribution of the rehabilitation to the direct use value of the wetland for reed harvesting. This is a conservative estimate of the monetary value of reed harvesting based on the local price of a bundle of reeds. Reeds are used to make grass mats, the majority of which are sold for cash income. If the value of reed mat production is used, rather than the price of reed bundles, the value of the contribution of the rehabilitation increases to R24 650 (assuming no wastage in converting reeds to reed mats).

Table 4.4: Contribution of the rehabilitation to the annual direct-use value of the Manalana wetland for reed harvesting, 2015

Measure	With rehabilitation (2015)	Without rehabilitation (projected)	Data source
Number of fields	34	8.5	Field survey/projected
% fields harvested	70	70	Field survey
Area reeds/field (m^2)	40	40	Field survey
% area harvested/field	70	70	Field survey
Area of reeds harvested (m^2)	666	167	Calculation
Bundle reeds/unit area (m^2)	1	1	Field survey
Number of bundles produced	666	167	Calculation
Wastage / non-use (%)	20	20	Assumption
Price/bundle (R)	R30	R30	Field survey
Value of production (R/year)	15 994	3 998	Calculation
Rehabilitation contribution (R/year)		**11 995**	

Note: All monetary values are 2015 Rands; discrepancies in sub-totals are due to rounding.

4.4.3 Livestock grazing

The wetland is used to graze livestock (cattle) during the winter dry season, typically a period of four to 10 weeks when livestock rely particularly heavily on the wetland for fodder. Participants of the focus groups and interviews explained that without the wetland grazing, cattle would lose condition, and some would die during the winter period. During the rest of the year crop cultivation is prioritized in the wetland over livestock grazing and fences are used to discourage cattle from the wetland.

Residents indicated that cattle theft had increased relative to the pre-rehabilitation surveys and particularly since 2010. Several focus group participants had lost some or even all their cattle to theft and many had chosen not to replace the animals; others had decided to sell their cattle before they were stolen. Participants agreed that cattle numbers had declined significantly over the last few years, but indicated that they weren’t sure of the actual total number remaining. In a separate interview, an elderly retired cattle farmer suggested that numbers had declined to a quarter of what they were five years previously and that only four households in Craigieburn continued to own cattle. The post-rehabilitation site visit took place in July 2015 during which two groups of cattle were observed grazing the wetland - a group of 10 and a smaller group (5 to 10 animals), indicating that the wetland was still being used for dry season grazing, but by significantly fewer numbers than reported from the pre-rehabilitation surveys (91 cattle, owned by 19 households). Ngetar (2011) reported an average of 25 cattle grazing in the catchment, which corresponds with the observation of the retired livestock owner.

In terms of the supply of grazing fodder, several features of the wetland are notable. The wetland is physically accessible to livestock; there is a moderate abundance of relatively palatable species (e.g., *Paspalum dilitatum* and *Leersia hexandra*); and the non-wetland areas within the catchment are in relatively poor condition, with a low cover of herbaceous vegetation. Ecological assessments of the extent and condition of the wetland during the post-rehabilitation fieldwork indicated that there had been no significant change in the area of the wetland vegetation nor in the species and quality of the vegetation since the pre-rehabilitation assessments and, therefore, no decline in the potential of the wetland to provide dry season fodder.

The value calculation and results are reported in Table 4.5. The number of cattle that could be sustained through the critical dry period in the 'with rehabilitation' case is 14, this is within the estimated number of cattle remaining in the area, suggesting that the wetland fodder is likely to be fully utilized (i.e., there is a demand for the potential supply). In the unrehabilitated case, the capacity of the wetland to provide dry season grazing is projected to be limited to four cattle (LSUs). The contribution of the rehabilitation to the direct use value of the wetland for grazing was estimated to R12 110 per year (i.e., the value of the additional 10 cattle sustained). However, should the number of cattle being kept in the area reduce further in future (i.e., to below 14 cattle), then the value of the rehabilitation with respect to the grazing benefit would decline.

Table 4.5: Contribution of the rehabilitation to the annual direct-use value of the Manalana wetland for livestock grazing, 2015

Measure	With rehabilitation (2015)	Without rehabilitation (projected)	Data source
Wetland area (ha)	3.5	0.9	Field Survey
Fodder available (kg/ha/year)	1 235	1 235	Related study
Total fodder available (kg)	4 323	1081	Calculated/projected
Fodder consumption (kg/LSU/day)	10	10	Assumption
Critical dry period (days)	30	30	Assumption
Number of cattle (LSUs) that could be sustained	14	4	Calculation
Direct use value to cattle owning household (R/LSU/year)	1 211	1 211	Related study
Value of livestock sustained (R/year)	16 954	4 844	Calculation
Rehabilitation contribution (R/year)		**12 110**	

Note: All monetary values are 2015 Rands; discrepancies in sub-totals are due to rounding.

4.4.4 Cost-benefit analysis

The CBA results, Table 4.6, indicate that the wetland rehabilitation generates an economic efficiency gain under both a 3% and 6% discount rate for a realistic 25-year lifespan, but not under the more conventional 8% discount rate. Under a longer timeframe, the benefits relative to the costs increase further for the 3% and 6% discount rates, with benefits exceeding costs two-fold at a 3% discount, and falling just short of a positive net present value (NPV) under an 8% discount rate. The fact that the benefit-cost ratios lie near to one under the 8% discount rate suggest that the results are likely to be sensitive to small changes in the benefit and cost estimates.

If the contribution to madumbe cultivation is taken as the sole benefit, a positive NPV is achieved only under a 3% discount rate. While the reed harvesting and livestock grazing benefits are relatively smaller than the madumbe cultivation benefit, their inclusion influences the CBA outcomes. This observation is important in that it highlights the sensitivity of the results to the inclusion of additional benefits. Including additional benefits, even if their monetary value is relatively smaller than the madumbe cultivation benefit, improves the economic efficiency outcomes. On the other hand, it indicates that the CBA results would be influenced by the falling away of any benefits over time, such as further declines in the use of the wetland for livestock grazing.

Table 4.6: Summary of the CBA results for the Manalana wetland rehabilitation based on the direct use value of the contribution of the rehabilitation to three livelihood activities, 2015

Rehabilitation contribution (annual)	**R89 079**
Madumbe cultivation	R64 974
Reed harvesting	R11 995
Grazing for livestock	R12 110
Cost of rehabilitation (upfront investment)	**R1 098 404**
Benefit-cost ratio	
25-year lifespan of structure	
3% discount rate	1.41
6% discount rate	1.04
8% discount rate	0.87
50-year lifespan of structure	
3% discount rate	2.09
6% discount rate	1.28
8% discount rate	0.99

Note: All monetary values are 2015 Rands; discrepancies in sub-totals are due to rounding.

Similarly, the results are sensitive to changes in the value of the benefits. This is illustrated in an example whereby the reed harvesting benefit is valued in terms of the final product produced, reed mats, instead of the price of reed bundles. Taking this approach generates a positive NPV under all scenarios except an 8% discount rate at a 25-year timeframe where the benefit-cost ratio falls just below one (0.99). Applying a replacement cost approach to value the livestock grazing benefit, using the average market price for grass hay, reduces the annual rehabilitation benefit from R12 110 to R3 058 and results in a weakening of the benefit-cost ratios.

In this case study, the benefits of the rehabilitation were estimated based on the difference in production under the 'with' and 'without' rehabilitation states – a marginal valuation approach. If the total wetland value post-rehabilitation is taken as the rehabilitation benefit, positive NPVs result under all three discount rates and both timeframes, with the highest benefit-cost ratio of 2.8 (3%, 50-year lifespan) and the lowest of 1.16 (8%, 25-year lifespan). At the recommended 8% discount rate, the difference between taking a marginal and a total value approach leads to different conclusions regarding the ratio of benefit to cost of the rehabilitation.

Given that the benefits reflect only the direct use value of three wetland-based livelihood activities, and that the benefit-cost ratios fall just below one for the 8% discount rate scenario, the results provide considerable justification for the investment in the rehabilitation from an economic efficiency perspective. This is further supported by the specific context of the wetland, that of a direct contribution of the wetland to the livelihood activities of vulnerable households and the fundamental role the rehabilitation structures play in securing the capacity of the wetland to provide these benefits[59].

On the other hand, the cost estimate is also partial, in that it does not include the costs associated with planning and co-ordination nor long-term costs of monitoring and maintenance of the wetland and the structures[60]. If an annual maintenance cost is included in the analysis based on a 2.5% of investment cost approach following de Groot et al. (2013), a positive NPV is achieved only under a 3% discount rate. What is appropriate in terms of monitoring and maintenance for the Manalana wetland is a question that remains to be addressed.

4.5 DISCUSSION

While the CBA results provide an indication of the economic efficiency gain achieved through the rehabilitation; the results considered alone do not provide a sufficient basis for drawing a conclusion on the value of the rehabilitation and whether the costs incurred are likely justified. Several aspects warrant further interrogation in interpreting the value estimates and CBA results. In addition, the limitations and uncertainties of the study require clarification.

4.5.1 Contribution as a proportion of household income

The most recent available census data (Stats SA, 2012) for the Bushbuckridge Municipality, suggest that 26% of households survive on an income of R4 800 or less per year (17% of which are estimated to have no income). A further 15% have an annual income of between R4 800

[59] There is no indication that the function of the structures (i.e., erosion control) could, in the foreseeable future, be replaced by natural processes within the wetland such as revegetation (Walters and Browne, 2018). The wetland is inherently vulnerable to erosion as a result of the physical attributes of the system, particularly the geomorphological setting. The rehabilitation efforts have 'suspended' this process rather than addressing the causes or 'drivers' of erosion.

[60] At the time of the study no formal monitoring and maintenance plan was in place.

and R9 600. Although the average annual household income was estimated to be R36 569, the greatest number of households fall within an income bracket of R9 601 to R19 600 (22%). At the time of the census, there were 34 069 recipients (6%) of old-age grants in Bushbuckridge (Bushbuckridge Municipality, 2014).

The focus group discussions identified the primary sources of income of the wetland users as pension grants, the sale of reeds or reed mats, the sale of wood collected from nearby old plantations, and sewing. Annual income from a pension grant (based on the 2015 grant value) amounts to R16 920. Often, and in the case of the Manalana wetland users, a single pension grant supports a number of dependents. The average household size for the Bushbuckridge Municipality was estimated as four during the 2011 census. For Craigieburn households, this is likely higher as average household size in 2004 was seven (Pollard et al., 2005). Based on an average household size of four, a single pension grant distributed across a household (R4 230 per person per year) falls far below the upper-bound poverty line for South Africa for 2015 of R11 904 per person year (Stats SA, 2017); even the average household income across the Bushbuckridge Municipality falls below this poverty line[61]. Among other categories, 'females', 'people from rural areas' and 'those living in Limpopo' are recognized as experiencing severe poverty challenges (Stats SA, 2017). It is clear that the residents of Craigieburn Village are vulnerable to extreme poverty.

It is against this backdrop, that the value of the contribution of the rehabilitation to local livelihoods is compared. At the time of this study, 28 women, predominantly elderly, were practicing wetland-based livelihood activities. Crop cultivation was the most widely practiced (all households), followed by reed harvesting (approximately 70%); while only five residents continued to own cattle.

Assuming benefits accrue equally across wetland farmers, the annual value of madumbe production per wetland farming household is R3 094[62]. Relative to a scenario of continued

[61] A caveat of this latter comparison to be noted is the different time periods of the estimates: 2011 average household income and a 2015 poverty line estimate.
[62] The returns of wetland cultivation are not distributed evenly as some households cultivate larger areas than others.

wetland degradation, R2 320 of this value can be attributed to the contribution of the rehabilitation. If compared with an annual income based on a pension grant, the contribution of the rehabilitation adds another 13% to the 'income' of the household. For those households falling into a lower income bracket of R4 800 to R9 600 per year, the value of the contribution of the rehabilitation is even more significant, equating to 24% to 48% of annual income.

While only a relatively small proportion of cash income is generated from wetland cultivation, the savings provided, as a source of food, are significant. Arguably, the contribution is greater even than a cost saving, in that few households could afford to substitute the cultivated produce through purchases. The wetland, and its rehabilitation, provide a food security and nutritional 'safety-net', the value of which is not captured or reflected in this market price valuation[63]. In the focus groups, residents spoke of 'hunger' and a need to provide food quantity and diversity to their families as primary drivers of wetland cultivation. Wetland cultivation 'buffers' households from food insecurity. Furthermore, the food security benefit reaches beyond the wetland farmers to the broader community of Craigieburn through the sale of produce at local prices which provides access to food that would otherwise not be available or affordable (local prices are cheaper than market prices and local purchases don't incur transport costs).

While the monetary value of reed harvesting is lower relative to crop cultivation, the sale of reed mats is regarded as an important source of income for those with few alternatives. In terms of the bundles of reeds harvested, the value of the contribution of the rehabilitation was estimated to be just under R12 000, or approximately R600 per household assuming 70% of wetland farmers harvest reeds[64]. This translates into a gross value of read mats of R24 650, or approximately R1 232 per household per year, without taking potential wastage into account. For those households falling into the lower income bracket, this is a meaningful contribution at 13% to 25% of average annual income. Discussions during the focus group

[63]Furthermore, local market prices were applied in the valuation which are lower than the cost of obtaining produce from elsewhere and the valuation was limited to madumbe cultivation due to lack of information on the production of other crops in the wetland and does not capture the added value of the extended growing season provided by wetland for the cultivation of other crops relative to dry-land production.

[64] The household surveys conducted prior to the rehabilitation indicated that 70% of wetland farmers harvest reeds (Pollard et al., 2005).

sessions and farmer interviews indicated that mats are prioritized for sale, with a few mats used for household purposes (generally those mats that have not been sold during the season). While the primary value of wetland crop cultivation relates to access to food and the associated nutritional benefits, reed harvesting and the sale of mats is an important source of cash income for households, particularly for those with few other sources of income.

The contribution of the rehabilitation in supporting livestock grazing during the dry season amounts to the capacity to sustain an additional 10 cattle relative to four cattle projected under a scenario of continued degradation; a direct use value of just over R12 000. From the discussions held during the site visit, it was apparent that cattle ownership within the area had declined significantly over the previous five years and that only five households continue to own cattle[65]. The value of the contribution of the rehabilitation equates to R2 400 per household per year for these cattle-owning households; 14% of household income based on a pension grant. However, cattle owning households tend to be wealthier and household income is likely to fall within a higher income bracket. The estimated value per household would be much lower in the case of a larger number of households owning cattle as the benefit associated with the rehabilitation (i.e., the additional number of animals that could be sustained) would be more widely distributed. The benefit is limited to the quantity of fodder the wetland is able to provide. The direct (consumptive) use values of the rehabilitation as a proportion of household income are summarised in Table 4.7.

Table 4.7: Direct use value of the Manalana wetland rehabilitation as a proportion of household income, 2015

	No. households	Pension grant income (R16 920 / year)	Lower income bracket (R4 800 – R9 600 / year)
Madumbe crop production	28	13%	24% to 48 %
Reed harvesting & mat sales	20	7%	13% to 25%
Livestock grazing	5	14%	*(unlikely to own cattle)*

[65] While the number of cattle kept in the area has declined significantly, observations and discussions with local residents suggest that the number still exceeds 14, suggesting that the additional capacity provided by the rehabilitation would still be utilized and therefore have an economic consumptive use value.

4.5.2 Comparison to other studies

There are relatively few published studies of the contribution of wetland rehabilitation to the direct use value of wetlands associated with local livelihood activities (Chapter 3). There are, however, a number of studies of the livelihood support value of wetlands, and several local studies of the economic value of wetland provisioning services. In South Africa, Lannas and Turpie (2009) and Adekola et al. (2012) analysed the value of livelihood benefits derived from the provisioning services of wetlands in an urban informal settlement (Western Cape) and rural (Limpopo) context, respectively. A comparison of results across selected studies is presented in Table 4.8, based on the *total value*[66] of each activity. Value estimates from the literature were converted to South African Rands (ZAR) and adjusted to 2015 prices for comparison.

In the study by Adekola et al. (2012), crop cultivation contributed the highest gross value, while sedge collection for mats generated the highest cash income[67]. In the study by Lannas and Turpie (2009), livestock grazing was the primary wetland-related activity, with some patches of the wetland used for crop cultivation. In the Manalana wetland case (this study), the highest gross value was associated with crop cultivation, yet the value of livestock grazing per household was highest due to the small number of livestock owners. Reed harvesting had the lowest gross value and value per household, but was an important source of cash income. This was similar to the findings by Adekola et al. (2012) who also noted the importance of reed harvesting as a source of cash income.

The role of wetlands in household food provision through cultivation was noted as an important contribution across all the studies and is commensurate with the findings of other studies of African wetlands (Schuyt, 2005). Compared to the two studies of South African wetlands, the value of crop cultivation per household for the Manalana wetland the

[66] For the Manalana wetland (this study), the total value of production/resources for the post-rehabilitation case was compared to the literature findings which related to the total value of production.

[67] In addition to sedge collection (for crafting mats) and cultivation, the wetland was used for livestock grazing reed collection (for building material), fishing, hunting, fuelwood collection, wild edible plant collection, medicinal plant collection and collection of water for drinking, washing and bathing (Adekola et al., 2012). While livestock grazing was valued, the authors considered the estimate too uncertain for inclusion in the overall wetland value.

corresponding proportion of household income, falls between that estimated by Lannas and Turpie (2009) and Adekola et al. (2012), Table 4.8.

Table 4.8: Comparison of direct-use value estimates for selected wetland resources, annual value per household and proportion of household income

	This study (2015)	Adekola et al. (2012)	Lannas and Turpie (2009)	Crookes (2003)		Nabahungu and Visser (2011)	
Location	South Africa, Limpopo	South Africa, Limpopo	South Africa, Western Cape	South Africa, Limpopo (Manganeng)	South Africa, Limpopo (Makua)	Rwanda, Nyagatare District	Rwanda, Muhanga District
Wetland specific	Yes	Yes	Yes	No	No	Yes	Yes
Size (ha)	9	100	310	n/a	n/a	1 080	225
Setting	Peri-urban – rural suburb	Rural	Urban - informal	Rural	Rural	Rural	Rural
Cultivation							
R/hh/year	3 094	4 666	1 146	3 503	4 693	21 230	938
% income	18%	26%	4%	10%	14%	74%	24%
Reeds (mats)							
R/hh/year	1 699	1 266	*n/a*	*n/a*	*n/a*	*n/a*	*n/a*
% income	10%	7%	*n/a*	*n/a*	*n/a*	*n/a*	*n/a*
Livestock							
R/hh/year	3 391	3 102[a]	45 266[b]	2 375	1 565	*n/a*	*n/a*
% income	20%	17%	90%	7%	5%	*n/a*	*n/a*

Note: Values extracted from the literature were converted to Rands and adjusted to 2015 prices for comparison; hh is household; [a] Noted by authors as a low confidence estimate; [b] Noted by authors as being very high.

Based on the setting of each wetland, the comparison seems to suggest that wetland cultivation values and the corresponding proportion of household income increase with a more rural setting. This makes sense, in that households closer to urban centres may have better access to a wider range of income generating opportunities and also that the scale of production may be limited in an urban context due to space constraints. However, as explained by Lannas and Turpie (2009), urban wetlands can perform a safety-net function in providing livelihood opportunities to vulnerable groups such as recent migrants seeking jobs as was the case in the urban Western Cape setting. The influence of scale is also demonstrated in the study by Nabahungu and Visser (2011) who attribute the differences in the value of wetland cropping and the relative proportion of income between the two Rwandan case studies to the small field size of the Muhanga district wetland as a result of land shortage. The authors note further the use of fertilizer in the case of the Nyagatare District wetland related to the relatively large size of the fields.

The estimated value of wetland reeds in terms of their value as mats and the corresponding proportion of household income compares well with the findings of Adekola et al. (2012). Wetland reeds for mat making were not identified in any of the other studies. Several site-specific factors are likely to influence the harvesting of wetland reeds. Firstly, is the presence of, and access to, reed species suitable for craft purposes. In the case of the study by Adekola et al. (2012) a decline in reed harvesting was noted and attributed to the conversion of remaining natural vegetation into cropping areas. The second important factor pertains to the demand for reed mats. In the case of the Manalana wetland context, an important source of demand stems from the use of reed mats in cultural initiation activities; participants explained that reed mats are bought by those coming into the area to participate in cultural activities. Without this demand, the value of wetland reeds would be much lower unless an alternative market for reed mats developed.

Valuing the contribution of wetlands to livestock production is in general, less straightforward relative to other wetland resources especially those that are traded directly in a market. This uncertainty is emphasised by the differences in the values and proportion of income reported across the studies (Table 4.8), with the proportion of household income ranging from 17% to 90% of household income for livestock owners. A key factor affecting the value of the wetland for livestock grazing is the proportion of total grazing needs derived from the wetland and the degree of dependence on the wetland (i.e., the availability of alternatives and the associated costs to access these). In many cases, livestock graze both dry-land and wetland areas and, as in this case, wetlands are often only grazed in the dry season (Shackleton et al., 2008). As such, the wetland may contribute a relatively small proportion to overall grazing needs, however, the timing of the contribution may be critical. In the case of the urban wetland studied by Lannas and Turpie (2009), the situation was different; the amount of dryland was limited and livestock owners relied heavily on the wetland for grazing throughout the year, which explains the high proportion of income attributed to the wetland (90%).

Common across the studies, was the conclusion that the wetlands contributed significantly to local livelihoods for both household food security and income generation. While the cash income derived from wetland use tends to be smaller than the household consumption value, this may be a critical resource for vulnerable households with few alternatives. After pension grants, the sale of reed mats was ranked at the next highest source of income for the Manalana wetland users. Similarly, Adekola et al. (2012) found wetland vegetation for mat-making to contribute the highest proportion to total cash income derived from the wetland and noted that wetland use to derive cash income often represents an economic safety-net for some members of society in that there are limited similar alternative income-generating options.

4.5.3 Access and alternatives

The direct use value of wetland rehabilitation derives from the intersection of supply and demand of wetland resources and services. People must be able to derive benefit from the additional potential of the wetland to supply resources and services due to the rehabilitation. The ability of households to derive benefits from ecosystems - termed 'access' - encompasses legal or sanctioned access to a resource as well as a range of additional influencing factors such as available technology; financial, human and social capital; knowledge; the physical setting; sense of identity and cultural influences; and social relations (Ribot and Peluso, 2003). Together, these factors shape people's abilities to benefit from wetlands and lead to differences in realized benefits across people or groups.

Land tenure within the Manalana catchment is based on traditional community arrangements. Focus group discussions revealed that access to agricultural land is simple and free; farmers do not have to pay for access. Wetland 'ownership' arrangements are relatively informal with agreement between neighbouring families being the key factor to 'owning' or using a portion of the wetland. Farmers explained that previous 'owners' and their family had first right to a wetland field, but mixed use/ownership arrangements could be negotiated. Generally, the extent of the area a farmer (or household) could 'own' depends on the size they are physically able to work. Focus group participants indicated that there was still land available for wetland farming, with options to establish new fields or adopt abandoned fields. Tenure of 'upland' or dryland parcels is more formalized with permission and 'papers' required from the Induna (traditional chief). Wetland farmers attributed the discrepancy between the two tenure processes to 'dryland' having more than one use and so requiring more attention and administration from the local authority.

The physical attributes of the wetland are such that the wetland is accessible to both humans and livestock. Residences are located within 400 metres of the wetland, the slopes adjacent to the intact portions of the wetland are gentle and there are several footpaths leading to, and around, the wetland. Species of reeds suitable for weaving are present and relatively abundant and reed patches are accessible through the wetland fields.

Labour appears to be an emerging factor of access to wetland cultivation in the Manalana wetland setting. With a decline in the participation of the youth (younger members of the household) in farming activities, some farmers indicated they are beginning to hire labour particularly for field preparation and fencing. Labourers are remunerated in cash, food or seedlings. Related to the decline of youth participation, is a loss of 'the knowledge of farming' described by the focus group participants, which may limit the ability of households to derive benefit from the wetland in future. Discussion of the reasons why the youth are 'no longer interested in farming' revealed possible social-cultural identity influences, with youth participants describing farming as 'old people's work' and suggesting there is a sense of embarrassment or shame associated with younger people (both men and women) working in the fields. There may be an element of socio-economic stigma underlying this sense of embarrassment, but this was not raised in any of the discussions. Other participants explained that children no longer have the time to assist their mothers and grandmothers with farming activities due to school commitments and as such are no longer familiar with farming as they grow older.

The livelihood support value of the rehabilitation of the wetland is mediated by the availability of alternative livelihood sources and substitutes for wetland-based activities and resources. Crop cultivation is the dominant wetland-based activity and provides income and food security benefits to farming households and food security benefits to the wider community of Craigieburn. There are other wetlands in the Manalana and adjacent catchments to which the farmers could gain access through the same tenure process; yet these wetlands are further away, in various states of ecological health, and are farmed by neighbouring villages. They are not a direct substitute for the Manalana wetland for the current farmers, but could provide an alternative option, albeit at a greater cost (e.g., physical effort, time, arising conflict with an influx of 'new' farmers to neighbouring wetlands).

The main reason given for farmers to take up wetland cultivation was to provide food for their families and generate income for necessities. For many of the farmers, this was at a time preceding the social grant programme. While in other parts of South Africa, access to grants has reduced the reliance on wetlands for food and as a livelihood activity (Hay et al., 2014), this does not yet seem the case for the Manalana wetland farmers, many of whom receive pension and or/child grants, but continue to farm. The reasons for this warrant further exploration, but appear to be linked both with the need to support a number of dependents and provide for the education of children / grandchildren and with a sense of purpose and identity. Focus group participants described farming as 'part of life', 'part of their nature' and that they wouldn't want to do anything else.

Wetland cultivation provides food for consumption for farming households and supports their food security, but it also provides access to produce to the wider community at lower prices through the sale of excess produce by farming households. Village prices are lower than supermarket prices; supermarkets are also further away and require transport which is an additional cost. This means that substitution of wetland produce, where available, comes at a higher price and is, therefore, not necessarily accessible. This finding is supported by other wetland studies in Africa (Schuyt, 2005) and studies of other ecosystems (e.g., Delang, 2006).

Direct substitution, even at a higher price, is not always possible. This is often the case of wild foods which have to be substituted with domestic crops (e.g., Delang, 2006), but was also evident in this case, where, according to the focus group participants, alternative sources of madumbe are scarce. Farmers explained that if the wetland was no longer available, madumbe production could be replaced by additional maize cultivation in the dryland fields to substitute food quantity, but the nutritional value and satisfaction derived from wetland grown madumbe could not easily be replaced. Amadumbe (taro) is recognized as a nutritionally valuable crop especially in comparison to other staple crops such as maize (Temesgen and Retta, 2015). Furthermore, by nature of the higher moisture content, wetland cropping allows for a longer growing season as well as the cultivation of a wider variety of crops relative to dryland cultivation thus providing additional nutrition over longer periods of the year. Loss of wetland cultivation has further implications in periods of drought; wetland cultivation is less vulnerable to drought than dryland cultivation. Households that cultivate a range of foods across a variety of 'places' (drylands, wetlands and household gardens) are relatively more food secure.

For livestock grazing, alternative sources of grazing are available, particularly with the decline in cattle numbers in the area, and it is likely that fodder available elsewhere in the catchment would be able to support the remaining livestock through the dry period. However, this could mean that stock would need to graze further away from the residences which could further increase the risk of theft. Given the significant reduction in the number of cattle present in the area, it may be that the provision of dry season grazing by the wetland is not as critical as it was in the past; however, it was apparent from the site visit observations and discussions that the wetland is still being used for dry season grazing. Dependence on the wetland for dry season fodder provision requires further exploration.

Livestock provide a number of benefits. Livestock manure is still widely used as fertilizer, and a local market for manure appears to be emerging, although wetland farmers indicated they are still able to obtain manure for free. However, some of the benefits associated with livestock are being replaced with alternatives. Residents indicated that banks are replacing cattle as a vehicle for savings, albeit out of necessity rather than preference for many of the previous cattle owners who expressed 'a lost sense of wealth' associated with not owning cattle. Similarly, use of cattle for ploughing dryland fields is declining, replaced with mechanised modes.

Only a few particular types of reeds are used for weaving reed mats and these species are confined to wet areas. In-field discussions with some of the wetland users revealed that reeds can be harvested along the river and not only from the wetland, but that physical access to these areas is more difficult, which also explains why many non-wetland farmers prefer to purchase bundles of reeds from farmers who have harvested reeds more easily from around and within their fields. Reed harvesting and mat-making was identified as an important source of cash income and as a livelihood activity that is accessible to most woman as it requires few additional inputs or demanding physical work.

4.5.4 Dynamic use and option value

Direct use of the Manalana wetland is dynamic. While some residents use the wetland consistently as part of their livelihood activities, others turn to the wetland in response to a particular need or event often related to a time of stress or household shock (e.g., job loss, family death). One resident told the story of having had no interest in farming when she moved to Craigieburn, but later turned to wetland cropping as a means to provide food for her family. She stopped farming once the period of stress had passed, but later turned back to wetland cropping when hunger again became a problem and out of a need to generate income to support her children's education. Several older wetland farmers indicated that they had returned to wetland farming in response to needing to provide for their grandchildren. For a younger woman, wetland cropping had become appealing after her job in a nearby town ended and she realised she could earn money through selling vegetables.

The focus group discussions revealed the importance to local households of having the option to use the wetland if they were not currently doing so, or for their children. The farmers explained that they retained their wetland fields and plots even if they didn't need to plant them all or use them every year. They preferred not to abandon their fields in case they wanted to use them at a later stage, or for their children / families to use in future.

The trend is similar for reed harvesting and mat-making which participants explained depends on the need for income at the time. Some users generally produce and sell mats each year; for others, selling of reed mats is undertaken only at certain times to generate income for a specific need (e.g., new school uniforms) or as a result of a household shock. There is also sporadic use of the wetland to access water[68] largely in response to failures or unreliability of municipal water service delivery. Direct use value estimates are derived from current use and,

[68] In this case, water supply is not necessarily a direct benefit of the wetland (and the rehabilitation) due to the presence of the stream. However, the wetland facilitates the collection of water through pools and gentle gradients. It was clear from the site observations that the gabion structure has contributed to this by providing an easy access point for water collection.

alone, fail to articulate the importance of having the option of using the wetland in future if needed.

The ‘option value’ dimension of the TEV typology provides a framing for the value of retaining the option to use or benefit from the wetland at some point in the future. While it is clear that the existing and past users of the Manalana wetland place value on the option to use the wetland in future, evident in their efforts to retain currently ‘unused’ fields; there is also an element of ‘unrecognized’ future use value for those who have not yet used the wetland, but may need to do so in future (this unknown future use is sometimes referred to as ‘quasi-option’ value).

Option value is a somewhat contested dimension of TEV with various arguments as to where it ‘fits’ in the TEV typology and how it can be measured. Hanley and Barbier (2009) suggest that ‘option price’, which reflects the willingness of current or future users to pay to secure the potential future use of the resource under conditions of supply or demand uncertainty, is the ‘best’ indication of option value. Stated preference valuation methods are proposed as the most appropriate for measuring option value (or option price), but there remain a number of challenges to their application particularly in a developing country context (Christie et al., 2012). Eppink et al. (2014:559) noted that option value “is not a frequent subject in applied studies”, yet, for the residents of Craigieburn, having the option to use the wetland in future is clearly important. In this case, the value of the rehabilitation, articulated as a monetary estimate of direct use value, is underestimated in not reflecting option value.

Counter to option value is the possibility of a decline in direct consumptive uses of the wetland in future; a trend observed in other parts of South Africa (Hay et al., 2014). In the Manalana wetland case, focus group discussions indicated that there had been a decline in the number of wetland farmers over the preceding eight years, largely a result of retired farmers not being replaced by new younger farmers. This was attributed to a preference for paid jobs and a lack of interest in farming of the youth. Younger focus group participants indicated that they could depend on grants and their grandmothers for money and food and could earn some money by harvesting wood and building fences while they sought out job opportunities. While the lack of interest and participation in farming by the youth suggests that wetland use may decline in future, many of the existing farmers had described how they had only started farming once they had a family and needed food and were not necessarily interested in farming as young women. Customarily, wetland cultivation appears to be identified foremost with older women.

The traditional system of wetland cropping to support household food consumption appears to be shifting in the present setting. Although there has been a decline in the number of wetland farmers, focus group participants agreed that there hasn't been a concomitant decrease in production, but rather that several of the farmers are cropping larger areas. Along with this, there appears to be an increase in the sale of produce both to local non-farming households and to markets outside of Craigieburn Village. A shift seems to be taking place from one of a majority of households cropping primarily for household consumption, to fewer households growing more and selling to the rest of the community.

Under this emerging system, direct wetland benefits accrue to fewer households (wetland farmers), yet many households (non-farmers) benefit indirectly through access to produce that would otherwise not be available (in the case of madumbe) or affordable. Focus group participants indicated that all the residents of Craigieburn depend on the food produced from the wetland. Through local cultivation, non-farming households have access to fresh produce even if they do not have the skills, interest, time and/or ability to farm themselves; this is both a food security benefit, for those who cannot afford alternatives, and a net financial benefit in the form of savings from access to produce at lower prices. Additional benefits of this shift in cultivation system were identified, including the emergence of paid work opportunities for building fences and field preparation for young men and additional income brought into the local economy through the sale of excess production outside of the village. Wetland farmers cited a stronger sense of identity associated with being increasingly recognized as playing an important role in the community. However, the longevity of this system, as more of the older farmers retire, and the broader question of the long-term use of the wetland remains to be addressed.

Attention to the dynamic nature and long-term use of the wetland is particularly relevant in this case given that the rehabilitation structures remain critical to averting further erosion of the wetland[69] and to continue to do so, their functionality will need to be maintained over the long term (decades to centuries). In addition, it can be expected that the wetland itself will experience changing biophysical conditions[70] in future influencing the patterns and predictability of erosion and deposition processes (Walters and Browne, 2018). The implication being that there is a need for regular monitoring and maintenance of the rehabilitation structures and, likely, repairs and additional or new structures in the future. Effectively, additional investment over the long term is expected to maintain the current condition of the wetland. The question arises of whether this investment will continue to be justified, given that it is predominantly natural geomorphological processes driving the erosion, especially if the livelihood value of the wetland declines.

To better address this question, a more thorough investigation of the likely future use of, and dependence on, the wetland from a livelihood perspective is required as well as a widening of the scope of benefits and values considered (e.g., additional direct use values, indirect, option and non-use values). This should include the food security and safety-net value of the wetland to local households, the role of the wetland in water security and water regulation at a catchment scale, and the contribution of the wetland to national and global priorities (e.g., habitat conservation and carbon sequestration).

Economic valuation and the TEV framework, particularly if a monetary metric alone is applied, may not be adequate or appropriate to assess and articulate this range of benefits and values. As explained by Turner et al. (2008:46):

"There are other sets of values that are supplementary to total economic value. These represent the role of wetlands in natural systems. They include the value of services that stabilize natural systems and perform protective and supportive roles for economic systems… the aggregate TEV of a given ecosystem's services, or combinations of such systems at the landscape level, may not be equivalent to the total system value".

[69] There is no indication that the function of any of the structures (i.e., erosion control) could, in the foreseeable future, be replaced by natural processes within the wetland such as revegetation (Walters and Browne, 2018).

[70] Anticipated changes include further development of the catchment and more frequent and intensive rainfall events associated with global climate change effects (Walters and Browne, 2018).

The direct use values estimated in this case study reflect a static view of use at a single point in time; yet focus group discussions revealed the dynamic nature of wetland use, identified the importance of option value, and highlighted emerging and potential shifts in future wetland use. These aspects require further exploration, which raises the question of how dynamic demand, future use, and option values could be investigated and articulated in the context of wetland rehabilitation evaluation. Wolff et al. (2015) similarly conclude that temporal dynamics of demand require further attention in the context of ecosystem services. Participatory scenario building offers promising direction in this regard, along with deliberative mapping, deliberative valuation and time use study methods which may also assist in addressing temporal considerations (Dunford et al., 2018; Harrison et al., 2018).

4.5.5 Limitations and uncertainties

One of the original aspects of this study is that it applied a marginal valuation approach to evaluate the specific contribution of wetland rehabilitation to the livelihood support value of the wetland, in the South African context. A comparison of the ecological health and functioning of the wetland 'with' and 'without' rehabilitation was informed by available evidence from detailed biophysical studies of the wetland. Further to determining the difference in supply potential, the use of the wetland post-rehabilitation was explored and considered in relation to the results of an extensive pre-rehabilitation assessment. This was important for establishing whether there was a demand for the potential supply of wetland resources secured through the rehabilitation.

The case study was limited to the contribution of the rehabilitation to the direct use value of three livelihood related benefits articulated using a monetary metric. The focus group discussions, findings of the pre-rehabilitation surveys, biophysical assessments and existing literature base point to the presence of additional benefits (e.g., nutritional health, sense of identity, medicinal plant harvesting) and values (e.g., indirect, option, non-use, and ecological and relational dimensions). For example, participants explained that wetland farming is 'part of life'; it is part of their identity and the social fabric of the community (sense of identity benefits, relational values). Wetland farmers expressed a desire for the wetland to be available for future use (option value) to future generations (bequest value). The pre-rehabilitation household survey (Pollard et al., 2005) highlighted a range of additional crops grown in the wetland (additional nutritional value), while the ecosystem services assessment (Walters and Browne, 2018) identified the carbon storage service potential of the rehabilitated wetland (indirect use value).

While the focus of this evaluation was limited to a few direct benefits, it was aligned with the primary objective of rehabilitation. The presence of additional benefits and values suggests a higher value of the contribution of the rehabilitation than demonstrated by the monetary valuation of a few direct use benefits. Three particularly relevant benefits in this case require further investigation: the health benefits of access to the nutritionally superior madumbe crop (in comparison to maize) and a wider diversity of vegetables for a longer period of the year, for both farming and non-farming households; the safety-net value of the wetland in terms of buffering residents from falling into severe poverty; and the importance to the residents of having the option to use (rely on) the wetland in future if the need arises.

In defining the ‘without’ rehabilitation scenario, no assumption of the likely impact of continued erosion on the yield of cultivated crops and wetland resources was made. Wetland areas support higher crop and forage yields relative to dryland areas through their greater moisture and nutrient content. Pollard et al. (2005) suggested that continued erosion of the wetland would result in losses of nutrients and organic matter from the system and reduce the fertility of the wetland. The research by Riddell (2011) linked gully erosion to desiccation of parts of the wetland and indicated that in the absence of the rehabilitation structures the Manalana wetland would continue to ‘dry-out’. As such, it could be expected that crop, reed and fodder ‘yields’ of the wetland would decline as the wetland continued to erode. However, there have been no measurements of yields under such conditions and so no assumptions were made of lower yields under the degraded scenario. The estimated difference in ‘production’ between the ‘with’ and ‘without’ rehabilitation states is likely conservative. The significance of this is illustrated in the sensitivity of the cost-benefit results to small changes in value of the benefits (Section 4.4.4).

Time spent on in-field data collection was a constraint in this case study. This was partially ameliorated by the earlier pre-rehabilitation household surveys and associated studies which provided a comprehensive baseline and reduced the time needed to develop a rapport with local residents. Additional in-field data collection spread over the cropping cycle would have enhanced the reliability and depth of this valuation and provided the opportunity to: conduct a household survey; address uncertainties regarding the number of cattle and grazing patterns; investigate the cultivation of additional crops; measure and observe a greater number of wetland fields and at different times; establish household incomes and the number of dependents specific to Craigieburn residents; and explore future likely scenarios and trajectories of change in wetland use. Production yield estimates based on respondent recall at a single point in time has been highlighted as a short-coming (Rasmussen et al., 2016) which could be addressed through a site visit during harvesting periods and/or through interviews carried out at multiple times during the year.

There are uncertainties associated with the annual direct use value per animal applied in deriving a value for wetland grazing. While the per animal value was 'transferred' from a locally relevant study (the Sand River catchment) which improves confidence, it is likely that the estimate is somewhat outdated in that a number of benefits associated with cattle ownership are no longer as relevant in the Craigieburn context (e.g., as a form of savings and source of milk and draught power). The use of village prices in valuing crop produce may have underestimated this value of the rehabilitation, as alternative local supplies of madumbe appear scarce and market prices are higher and transport costs are incurred in obtaining produce from nearby markets.

4.6 CONCLUSION

This case study has attempted to measure the direct (consumptive) use value of the contribution of wetland rehabilitation to three wetland-based livelihood activities using a combination of market-price (crop cultivation and reed harvesting) and value-transfer methods (livestock grazing). Given that the assessment measured only some of the benefits and values associated with the rehabilitation, and that the benefit-cost ratios fall just below one under the higher discount rate scenario, the results provide considerable justification for the investment in the rehabilitation. This is further supported by the specific context of the wetland, that of a direct contribution of the wetland to the livelihood activities of vulnerable households and the fundamental role the rehabilitation structures continue to play in securing the capacity of the wetland to provide these benefits.

The proximity of the cost-benefit ratios to a value of one does suggest that the assessment is particularly sensitive to changes in the benefits and costs included and, or, the value measurements. As shown for the reed harvesting and livestock grazing benefits, applying different methods to measure the value of the consumptive use benefit produces different value estimates and, in this case, influences the cost-benefit outcomes and therefore the potential economic efficiency conclusion. Similarly, the CBA results would be influenced by the inclusion of additional benefits (and values), the falling away of any benefits over time (even relatively small changes such as further declines in the use of the wetland for livestock grazing), and the inclusion of additional costs (such as long term monitoring and maintenance).

Several of the findings point to the presence of additional benefits and values associated with the rehabilitation. One of the potential additional values identified through the case study is that of option value. Focus group discussions revealed the importance to local households of having the option to use the wetland in future, even if they weren't currently doing so, or for use by their children. This suggests that the value of the rehabilitation, articulated as a monetary estimate of direct use value, is underestimated in not reflecting the importance of having the option to use the wetland in future if needed.

The assessment has highlighted the importance of contextualising the value estimates with respect to the specific situation of the beneficiaries. While the estimated R2 320 per household per year value of crop cultivation may not seem much to many, for this beneficiary group it represents a significant proportion of annual income (13 to 48%) and a substantial cost saving. Arguably, the contribution is greater even than a cost saving, in that few households could afford to substitute the cultivated produce through purchases, and the rehabilitation is, therefore, also a contribution to food security (including nutritional health through food diversity). This 'saving' could mean the difference between falling into severe poverty for a household, or the need to give-up other essential aspects including education. In this case, the value of the rehabilitation extends far beyond the market value of the wetland resources to values associated with essential food security and safety-net functions for local households. Indeed, the food security value extends beyond the wetland farmers by way of providing a local, cheaper and more diverse source of vegetables for local households.

Several of the aspects considered and discussed in this chapter emphasise the importance of interrogating and interpreting the value estimates in respect of the specific context when communicating the findings and drawing a conclusion regarding the 'justification' of the rehabilitation. The case study further highlighted the dynamic nature of the wetland as a social-ecological system in respect of future potential changes in both the biophysical (supply) and social (demand) elements, whereas standard valuation methods reflect a static view and generate an estimate of value for a single point in time. Ways to investigate and articulate temporal dynamics and option value in the context of wetland rehabilitation requires further attention. Participatory scenario building offers promising direction in this regard, along with deliberative mapping, deliberative valuation and time use study methods.

The next chapter presents a case study of a pre-rehabiliation evaluation that illustrates the economic valuation of an anticipated improvement in the water quality regulatory service of the wetland as a result of the rehabilitation using the replacement cost method. Drawing from the experience of the case study application, the chapter further proposes a set of factors to consider when using the replacement cost method to improve the reliability of the results.

CHAPTER 5: MTHINZIMA WETLAND WATER QUALITY ENHANCEMENT: A COST-BENEFIT ANALYSIS

5.1 INTRODUCTION

This case study addressed the economic valuation of the anticipated gain in nitrogen retention potential as a result of the rehabilitation of a degraded channelled valley bottom wetland located strategically between a rapidly urbanizing settlement and an impoundment of the regionally important uMngeni water supply system[71]. The replacement cost method was applied, which is often considered a conceptually simpler approach; however, application demonstrated that the method can still be challenging to apply and relies on several assumptions which may affect the reliability of the results. From a reflection on the case study application, a set of eight factors to consider when using the replacement cost method in wetland rehabilitation valuation and decision-making have been derived.

The specific goal of the rehabilitation in this case stady was to enhance the water quality amelioration services of the wetland. Concerns have been raised regarding the risk of eutrophication of the impoundments within uMngeni water supply system (GroundTruth, 2012; Jogiat, 2013; Matthews and Bernard, 2015). Wetland systems are recognized as being effective in addressing non-point source water pollution, particularly the retention of nutrients through biogeochemical processes such as denitrification (Verhoeven et al., 2006, Marton et al., 2015). Their effectiveness in this regard is well-established in a large body of literature documenting studies of nitrogen and phosphorous removal by wetlands (see Land et al., 2016, for example). Indeed, wetland systems are utilized and constructed specifically for this purpose, in addition to playing an important role in maintaining (or improving) water quality naturally (Kadlec and Wallace, 2009; Mitsch and Gosselink, 2015). Given their capacity for nutrient assimilation, wetlands, and the rehabilitation of degraded wetlands, can be viewed as one component of efforts to counter eutrophication (Land et al., 2016).

[71] This case study was part of the Water Research Commission Project K5/2345 and preliminary parts of the analysis were reported in project deliverable 12 and the final technical report (Jewitt et al., 2020). The evaluation was undertaken by M Browne with contributions from N Buthelezi, S Ferrer and G Jewitt.

Many of South Africa's impoundments and freshwater resources exhibit high nutrient enrichment and are considered moderately to highly eutrophic (Oberholster and Ashton, 2008; Mathews and Bernard, 2015). This is of concern as eutrophication can have several negative impacts, including ecological impacts, in the form of biodiversity loss, aesthetic and recreational impacts and human health impacts (DWAF, 2002). Eutrophication of water storage impoundments compromises the quality of raw water treated for drinking water purposes and is associated with increased water treatment costs (Pretty et al., 2003; Graham et al., 2012; Umgeni Water, 2019). Addressing the eutrophication of freshwater resources is a priority concern for water security in South Africa (DWS, 2018); the foremost management action in this regard is to minimise the input of nutrients into water resource systems (van Ginkel, 2011).

Given its strategic location between a peri-urban settlement experiencing on-going sanitation issues and a water storage impoundment, the Mthinzima wetland is well positioned to intercept nutrients carried in the streamflow thereby reducing the load of nutrients to the impoundment and the associated eutrophication risk. However, the biophysical drivers of the wetland have been significantly impacted upon by historical and present activities affecting its capacity for nutrient assimilation (GroundTruth, 2015). Given this context, rehabilitation of the wetland was proposed with the primary objective of optimising the water quality enhancement services of the wetland. The aim of this case study assessment was to derive an economic value for the gain in nitrogen reduction potential of the wetland as a result of the rehabilitation and compare the added value to the cost of the rehabilitation. The intention was to demonstrate the potential benefit of investing in the rehabilitation of the wetland articulated through an economic value framing.

Several studies have attempted to estimate the economic value of the water quality enhancement benefits of wetland rehabilitation. The quantitative review of published studies on wetland rehabilitation valuation (Chapter 3) indicated that water quality related services were valued in almost a third of the studies. The majority of which applied the replacement cost method whereby the cost of an alternative measure to achieve the same level of water quality improvement is used as a 'proxy' value for the wetland water quality service. All these studies took as the replacement or alternative a form of engineered chemical wastewater treatment[72]. Two studies also considered measures to reduce nitrogen emissions from agriculture (Meyerhoff and Dehnhardt, 2007; Grossman, 2012) as an alternative. None of these studies, however, were for wetland rehabilitations in Africa. Through a meta-analysis of the value of wetlands in developing countries, not specific to rehabilitation, Chaikumbung et al. (2016) identified 81 sites of study in Africa (excl. North Africa), of which almost 20% valued the water treatment services of wetlands. No indication of the specific valuation method was given.

The economic value of wetlands in South Africa has been explored through several studies (e.g., van Zyl et al., 2004; Lannas and Turpie, 2009; Kleynhans et al., 2010; Scovronick and Turpie, 2010; Turpie et al., 2010; Maila et al., 2017). Of these studies, Turpie et al. (2010) and Maila et al. (2017) investigated the water quality enhancement services of wetlands and both applied the replacement cost approach based on the costs of standard wastewater treatment systems. While the case study by Maila et al. (2017) was specific to wetland rehabilitation, only the value of the water quality enhancement services of the wetland post-rehabilitation was estimated, rather than the value of the contribution of the rehabilitation to improving the services. Van Zyl et al. (2004) investigated the economic value of the specific contribution of wetland rehabilitation, but did not consider the water quality enhancement services of wetlands. All the water quality enhancement valuations focused on the nitrogen and phosphorus abatement services of wetland habitats, except for the study by Maila et al. (2017) which demonstrated the sulphate reduction capacity of wetlands in an acid mine drainage context, and emphasised the capacity and importance of wetlands for water quality enhancement and, particularly, for nutrient reduction.

This case study attempted to derive an ex ante estimate of the economic value of the expected additional nitrogen reduction service gained through wetland rehabilitation. This involved quantifying the additional nutrient retention potential of the wetland due to the proposed rehabilitation and then applying the replacement cost method to attribute an economic value to wetland nitrogen abatement. In a second step, the estimated value of the

[72] Eight studies - Meyerhoff and Dehnhardt (2007); Tong et al. (2007); Chen et al. (2009); Broekx et al. (2011); Lei et al. (2011); Grossman (2012); He et al. (2015); and Russell and Greening (2015).

gain in nutrient reduction as a result of the rehabilitation was compared to the projected cost of the rehabilitation intervention in a partial CBA. The case study description is structured as follows: the study area and rehabilitation context are introduced followed by a description of the methods and data collection. The focus is on the approach to quantifying, ex ante, the expected reduction in nitrogen load as a direct result of the rehabilitation and replacement cost method of ecosystem service valuation. The results of the valuation and the CBA are then presented followed by a detailed discussion of the uncertainties and limitations of the assessment and the implications for the results and for wetland ecosystem service valuation using the replacement cost method.

5.2 STUDY AREA AND REHABILITATION CONTEXT

The wetland of this case study lies along the Mthinzima Stream which falls within the catchment of a strategically important water supply system and is a tributary of Midmar Dam, a key water storage impoundment of the uMngeni water supply system and source of recreational value. The uMngeni water supply system supports over six million people, serving the large metropoles of Pietermaritzburg and eThekwini and surrounds in KwaZulu-Natal and providing water to South Africa's third largest regional economy. Midmar Dam is of significant recreational value to both local and international visitors and is the setting for the internationally recognized Midmar Mile swimming event and numerous other sporting events and activities. The Mthinzima Stream, and the wetland, provide drinking water access for livestock grazed by the adjacent Mpophomeni Community. In light of the sanitation management issues associated with the settlement within the Mthinzima catchment, the stream can also be viewed as a wastewater conveyance system, in terms of 'water uses', benefitting the local population and, arguably, the district municipality.

The Mthinzima Catchment contains areas of natural grassland and forest, but is dominated by the rapidly urbanizing settlement of Mpophomeni which is situated approximately four kilometres upstream of Midmar Dam. Mpophomeni was established in the late 1960s to early 1970s to provide housing for people forcibly removed from surrounding areas as part of the apartheid system (Baiyegunhi and Makwangudze, 2013; Denis, 2013). Mpophomeni has continued to expand since its establishment with both formal and informal housing and commercial developments (Umgeni Water, 2013; Rivers-Moore, 2016); the 2011 census puts the population at approximately 25 700, whereas a local NGO suggests the number is closer to 35 000 (Masibumbane, 2020). As for many similar settlements across South Africa, town planning and service delivery have been influenced by this political context. Mpophomeni is characterized by inadequate and dysfunctional sanitation infrastructure and poor provision of waste collection services (Terry, 2017; Umgeni Water, 2019).

The Mthinzima Stream receives pollution emanating from the settlement. There are two main sources of contaminated discharge to the Stream. At present, Mpophomeni does not have an operational wastewater treatment works; wastewater collected through the formal sanitation system is pumped to a treatment works in a nearby town. During this process, wastewater is collected and stored in ponds; due to natural seepage and excess flows, wastewater from these ponds leaks into the Mthinzima Stream. Moreover, the wastewater conveyance system is poorly designed, old, prone to storm water ingress and improperly used, resulting in regular overflows of sewage into the stream through surface run-off[73]. This has resulted in chronic pollution inputs to the Mthinzima Stream and compromised the water quality of the system (van Deventer, 2012; Ngubane, 2016) which has had a negative impact on the water quality of Midmar Dam (Umgeni Water, 2019).

Several studies and routine monitoring have documented diffuse pollution as a major contributor to the degradation of rivers and streams in the catchment and raised concerns that the impoundments of the catchment could become eutrophic (GroundTruth, 2012; Matthews and Bernard, 2015; Rivers-Moore. 2016; Namugize et al., 2018; Umgeni Water, 2019). Sewage effluent is a major source of nutrient inputs to aquatic systems in South Africa (DWAF, 1996; Matthews and Bernard, 2015). Results of water quality studies in the Mthinzima catchment have concluded that there has been an increase in nutrient loads entering Midmar Dam from the Mthinzima Stream over time (van Deventer, 2012; Ngubane, 2016) and shown that nutrient levels of the Mthinzima Stream influent to Midmar Dam

[73] The pipes of the waste water conveyance system are inadequate for the size of the community as well as being laid at incorrect gradients; misuse of the system relates largely to the disposal of solid waste into the sanitation system (Terry 2017; DUCT, 2018). At least 70 manholes within the Mpophomeni area regularly become blocked and surcharge (DUCT, 2018).

exceed the recommended eutrophication thresholds prescribed for South Africa (Namugize et al., 2018). A reduction in water pollution into Midmar Dam, notably nutrient inputs associated with sewage contamination of the Mthinzima Stream, has been identified as a priority management action (WRC, 2002; GroundTruth, 2012; Jogiat, 2013; Umgeni Water, 2019). To this end, two infrastructure investment projects have been proposed which aim to contribute to the reduction of nutrient inputs to Midmar Dam associated with sewage contamination of the Mthinzima Stream. The district municipality has planned a new Waste Water Treatment Works (WWTW) and associated treated effluent pipeline to service the Mpophomeni area which has received environmental authorization (Umgeni Water, 2013; KZN DAEA, 2014). As part of the 'Save the Midmar Dam' ecological infrastructure initiative[74], the rehabilitation of wetlands within the upper uMngeni catchment has been proposed (Jogiat, 2013; Umgeni Water, 2018) and a rehabilitation plan has been developed for the Mthinzima wetland (GroundTruth, 2015).

A third water pollution reduction initiative, the Mpophomeni Sanitation Education Project (MSEP) has been underway since 2011. The aim of the project being to reduce sewage and solid waste pollution and improve the lived environment and public health of the Mpophomeni community through increased education and awareness around the pollution of water and the monitoring of pollution incidents[75]. The initiative has been successful in reducing the incidence of long-term manhole spillages through enhanced community action, providing significant value to the Municipality at a relatively low cost (Duzi Umngeni Conservation Trust (DUCT), 2018).

5.2.1 Proposed wetland rehabilitation

There are several wetlands within the Mthinzima Stream and Mpophomeni area. Together these areas are known as the Mthinzima Stream Wetland Complex. The proposed rehabilitation of a 98 hectare portion of wetland habitat downstream of Mpophomeni - the Mthinzima wetland – is the focus of this case study. The Mthinzima wetland is situated on

[74] A partnership of local stakeholders including uMgungundlovu District Municipality, KZN Department of Environment and Traditional Affairs and local communities (Jogiat, 2013; Umgeni Water, 2018).

[75] The project has provided environmental, educational and monitoring services through the employment and capacity building of Enviro-Champs conducting activities including Door-to-Door visits, the monitoring and reporting of surcharging manholes and youth clubs (DUCT, 2018).

land owned by the Zenzele Trust that is used primarily for livestock (cattle) grazing. The rehabilitation plan was compiled in 2015 by GroundTruth consulting on behalf of the Zenzele Community Trust members[76].

According to the rehabilitation plan, the present condition of the wetland is classified as largely modified as a result of several negative impacts to the biophysical drivers of the wetland as a result of historical and ongoing activities including excavation of artificial drainage channels, intensive grazing by livestock and alteration to water flow and erosion in the system's catchments. The intention of the proposed rehabilitation is to address several of these issues with the aim of securing and improving the functioning of the system specifically to enhance the water quality services of the wetland (GroundTruth, 2015). The primary anticipated outcome of the rehabilitation is a reduction in nutrient loads to the impoundment. In this way, the rehabilitation intends to contribute to reducing the risks of contamination and eutrophication of the Midmar Dam water resource.

The planned rehabilitation incorporates site-specific interventions primarily intended to redistribute water flows across the surface of the wetland rather than being confined to the incised channel. The proposed interventions include the construction of concrete and earth structures, re-shaping earthworks and the infilling of artificial drains, and revegetation of disturbed areas. The rehabilitation plan also specifies long-term maintenance activities including the removal of excess plant material at regular intervals (burning or brush-cutting), control of emerging alien plants and monitoring and maintenance of the structural interventions.

5.2.2 Proposed waste water treatment works (WWTW)

The planned WWTW will be located adjacent to Mpophomeni and will cater for both Mpophomeni and a proposed new housing development. The WWTW has been specifically designed to reduce the risk of pollution to the nearby Midmar Dam. A requirement of the

[76] The delineation, assessment of the functioning and integrity of the wetland within the study site for current and post-rehabilitation scenarios, and the rehabilitation plan were undertaken in accordance with the general process adopted by the national Working for Wetlands programme and in accordance with the approach outlined in WET-RehabPlan (Kotze et al., 2009).

environmental authorization for the planned WWTW is that all treated effluent must be discharged downstream of Midmar Dam (KZN DAEA, 2014) and the infrastructure and associated effluent pipeline has been designed accordingly. Overflows from the maturation ponds associated with the works are considered unlikely due to the implementation of a storm overflow pond and the hybrid maturation river; in the event that overflows do occur they will have been treated and the effluent disinfected (KZN DAEA, 2014).

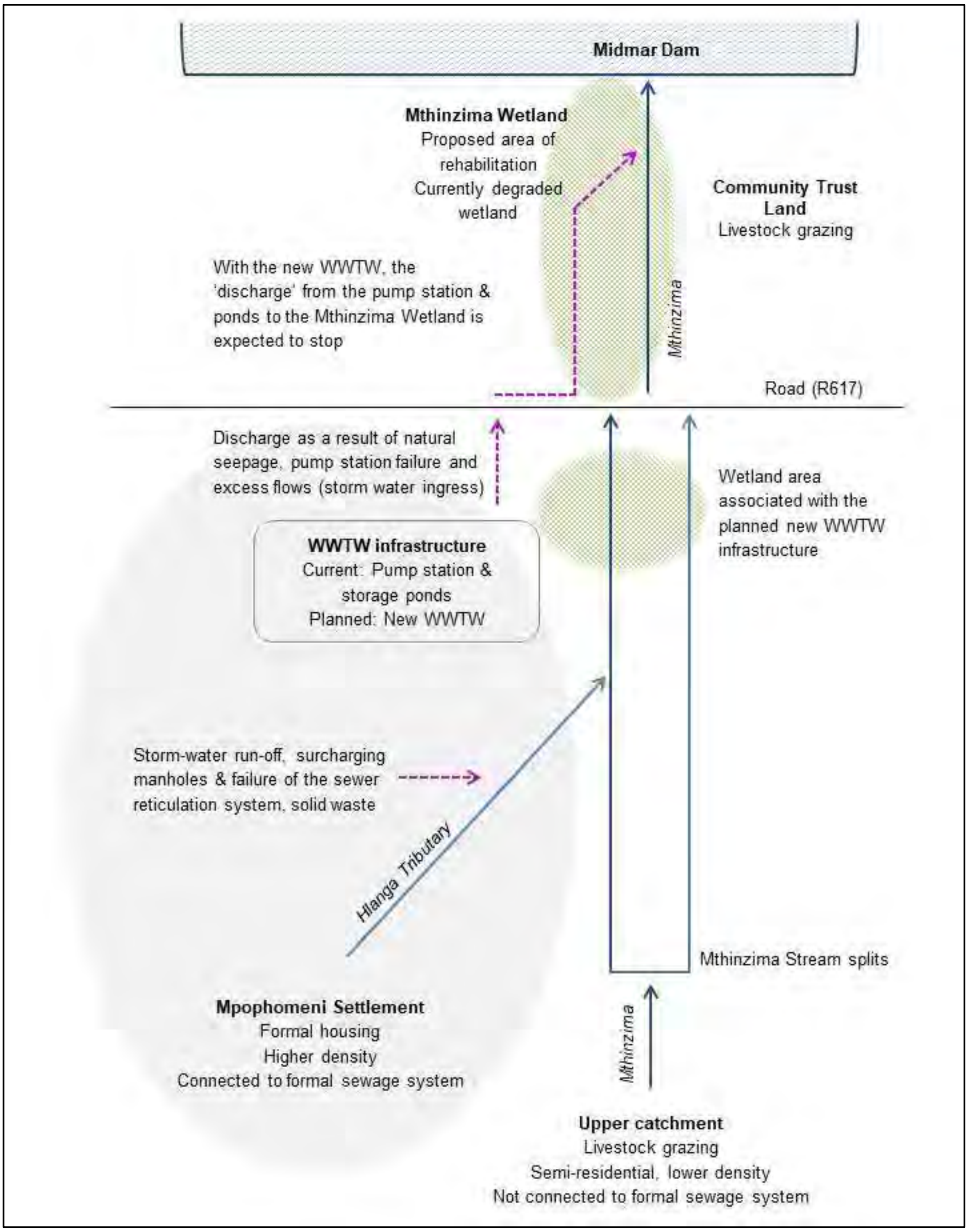
Included in the plans for the proposed WWTW is the refurbishment of the main sewer pipeline through Mpophomeni to the works. There are no plans to upgrade any further sections of the existing sewage conveyance network and it is anticipated that raw sewage contamination of the Mthinzima Stream from regular failures of the conveyance system will continue (Terry, 2017). Furthermore, there is some risk of untreated effluent spills during construction of the proposed WWTW and during the first year of operation (Terry, 2017).

5.2.3 System description

Figure 5.1 is a schematic illustration of the 'settlement-wetland-impoundment' system (the Mpophomeni Settlement – Mthinzima wetland - Midmar Dam system) highlighting the infrastructure, pollution inputs and water flows specifically relevant to this case study and the interactions between them. At present, there are two main sources of flow to the Mthinzima wetland:

1. The Mthinzima stream itself which is natural stream flow with inputs from storm water run-off from Mpophomeni, surcharging manholes and failure of the sewer reticulation system and solid waste; and
2. Frequent outflows from the wastewater pump station and pond infrastructure, as a result of pump station failure and storm water ingress, which have created a flow path from the wastewater ponds through the wetland area. The flow is a combination of natural seepage and semi-treated effluent.

With implementation of the proposed Mpophomeni WWTW, the 'flows' and associated pollution inputs originating from seepage and overflows of the existing waste water ponds (point 2 above) will stop. However, run-off from the Mpophomeni Settlement, which is characterized by pollution inputs from regular failure of the wastewater conveyance system and discharges not linked to the reticulated sewerage network, will continue to drain into the Mthinzima Stream.



Note: WWTW – wastewater treatment works.

Figure 5.1: Schematic orientation of the 'settlement-wetland-impoundment' system highlighting the relevant infrastructures, pollution inputs and water flows.

5.3 METHODS AND DATA

5.3.1 Overview

The object of this valuation case study is the additional nitrogen retention capacity gained as a result of the rehabilitation. While rehabilitation valuation should attempt to cover the broadest range of benefits, in this case the specific objective of the rehabilitation is to enhance the water quality related services of the wetland. The corresponding primary question of interest is whether the benefit of the enhanced nitrogen retention capacity of the wetland justifies the cost of the rehabilitation from an economic efficiency perspective. The economic value of the additional nitrogen retention capacity is a function of the extent (quantity) of the nitrogen retention potential gained and the demand for the additional capacity which stems from the demand for the benefit(s) associated with additional nitrogen retention. It follows, that to derive an economic value estimate for the additional nitrogen retention as a result of the rehabilitation requires first quantifying the degree to which nitrogen release into the downstream system is reduced as a result of the rehabilitation and then assigning an economic value to this reduction.

The expected gain in nitrogen retention capacity was quantified through a modified area-based calculation drawing on the concept of ‘hectare equivalents’, an approach applied in South African wetland rehabilitation practice as a metric of functional wetland habitat gained or lost under different scenarios (Cowden and Kotze, 2008). The primary benefit associated with increased nitrogen retention by the wetland is a reduction in the risk of eutrophication of the downstream Midmar Dam and avoidance of the associated negative impacts, such as increased water treatment costs and reduced recreational value related to eutrophic water. The direct estimation of these benefits would require modelling the relationship between nutrient input to the Dam and nutrient concentrations within the Dam itself and the associated eutrophication risk and relating this specifically to changes in nutrient inputs as a result of the rehabilitation of the Mthinzima wetland. A second relationship would need to be estimated relating the level of eutrophication to the demand for water for potable and recreational purposes.

Eutrophication is a complex issue (Thornton et al., 2013). Changes in nutrient inputs, eutrophication level and the subsequent consequences for water use are subject to an interdependent system of climatological, physical and social factors (Graham et al., 2012) resulting in relationships that are non-linear, subject to threshold effects and site-specific. There is no single definitive model for predicting changes in eutrophication risk with reductions in inflow nutrient loads. Modelling approaches all require a high level of knowledge specific to the individual water body to inform their application and interpretation (Thornton et al., 2013). In the initial stages of this study, it became clear that available knowledge of the internal biological and chemical functioning of the Midmar Dam to support the required level of modelling was inadequate (Jewitt et al., 2020). In the case of data and model limitations, the nitrogen retention potential of a wetland can be assigned an economic value through an indirect approach using the replacement cost method. An indirect approach is commonly applied in valuing wetland ecosystem services related to water quality improvements (Woodward and Wui, 2001; Chapter 3).

Lacking the data and model sophistication required for the direct approach, the indirect replacement cost method was applied in this case study. A key weakness of any cost-based valuation approach is that it fails to measure willingness to pay for the benefit (or willingness to accept compensation for a benefit loss) (Barbier et al., 1997; Heal et al., 2005). However, in this case study context, taking a cost-based approach is supported by evidence of the social demand for nitrogen reduction within the Mthinzima System (see Section 5.3.3). The estimated value of the additional nitrogen retention potential of the wetland with rehabilitation was compared to the projected cost of the rehabilitation within a partial cost–benefit analysis framework to assess whether the gain in nitrogen retention capacity of the rehabilitated wetland justifies the cost of the intervention based on an economic efficiency criterion.

For this case, the net present value of the wetland rehabilitation can be written as:

$$NPV_t = \sum_{t=0}^{T} \frac{1}{(1+r)^t} [VNR_t - CR_t]$$

where NPV is the net present value of the proposed wetland rehabilitation, VNR is the value of the additional nitrogen retention service and CR is the cost of the rehabilitation (investment and maintenance). Missing from the equation, and reflecting the partial nature of the analysis, are the benefits of a gain in other ecosystem services because of the rehabilitation and the opportunity cost of forgone alternative land use. In this case, the present land use is livestock grazing which is not expected to be entirely forgone as a result of the rehabilitation and the rehabilitation interventions have been designed to be resilient to livestock impacts.

5.3.2 Quantifying the gain in wetland nitrogen retention potential

Nitrogen reduction to the downstream system is one of the ecosystem services provided by wetlands (Turner et al. 2008; Barbier, 2011) and can be enhanced through rehabilitation interventions (Land et al., 2016). However, individual wetlands differ in their effectiveness in providing nitrogen reduction services according to characteristics such as the dominant source of water to the wetland (e.g., surface vs. sub-surface), how water flows through the wetland and exits the wetland, the geomorphic setting of the wetland and the extent of vegetation cover (Kotze et al., 2008; Mitsch and Gosselink, 2015). For example, many of the ecological processes contributing to nitrogen trapping and assimilation by wetlands, such as sedimentation, denitrification and plant uptake, are enhanced by slower flows of water through the wetland (longer residency time within the wetland) and greater contact surface of water with plants and sediments (contact with soil decomposers and decomposition processes) (Kotze et al., 2008; Kadlec and Wallace, 2009; Mitsch and Gosselink, 2015).

In the case of the Mthinzima wetland, various ecological assessments including an ecosystem services assessment comparing the with and without rehabilitation cases[77] indicated the

[77] Undertaken as part of the development of the rehabilitation plan (GroundTruth, 2015), following the WET-EcoServices assessment protocol (Kotze et al., 2008) and WET-Health assessment method (Macfarlane et al., 2008) adopted by the South African Working for Wetlands programme (a component of the Natural Resource Management Programme a state funded Expanded Public Works Programme).

potential of the proposed rehabilitation to lead to an increase the effectiveness of the wetland to provide ecosystem services associated with water quality enhancement (GroundTruth, 2015). Based on the ecological assessments of the degraded wetland, drivers of degradation within the system and the historical context, an increase in the effectiveness of the wetland to reduce stream flow nitrogen levels is expected as a result of the rehabilitation (GroundTruth, 2015). To estimate the level of gain in nitrogen reduction potential of the wetland because of the rehabilitation, and therefore the degree to which nitrogen release into the downstream system could be reduced, an 'area extent' approach was adopted whereby unit area nitrogen retention rates for restored wetlands were applied to the projected gain in area of wetland habitat with the rehabilitation. A comparable approach has been adopted in similar studies valuing the water quality enhancement services of restored wetlands (for example by Jenkins et al., 2010 and Russell and Greening, 2015).

To determine the likely gain in area of wetland habitat because of the rehabilitation, it is necessary to consider the influence of the rehabilitation efforts on the integrity of the wetland in addition to the spatial extent so as to approximate a gain in area of *functional wetland habitat* (Cowden and Kotze, 2008). The rationale behind the approach being that rehabilitation efforts could lead to a slight improvement in wetland integrity across a large area of degraded wetland or a significant improvement in integrity across a small area of the wetland, which may result in a similar change in ecosystem service potential. In South African wetland rehabilitation practice, the 'hectare equivalents' approach is used in assessing wetland intervention outcomes and provides a standard metric of the area of functional wetland habitat gained (or lost) derived from a rating of the ecological integrity of the wetland under different conditions or scenarios (Cowden and Kotze, 2008).

The area of functional wetland habitat for the present state and a projected 'with rehabilitation' state was determined as part of the wetland rehabilitation plan (GroundTruth, 2015). Table 5.1 indicates the integrity[78] of the wetland with rehabilitation (projected) and without rehabilitation (present state) and the corresponding area equivalent of functional

[78] Derived using the WET-Health assessment tool (Macfarlane et al., 2008) to calculate the Present Ecological State (PES) scores (GroundTruth, 2015).

wetland habitat. The present ecological state classes describe the integrity of the wetland, where ‘A’ reflects an “unmodified, natural” state and ‘F’ represents the state where “modifications have reached a critical level and the ecosystem processes have been modified completely with an almost complete loss of natural habitat and biota” (GroundTruth, 2015:19). See Box 1 for a full description of all classes.

Table 5.1: Summary of the ecological integrity (present ecological state class) and equivalent functional habitat of the Mthinzima wetland under the ‘current’ state and a projected ‘post-rehabilitation’ state, 2015

Characteristic	**Present Ecological State class**	
	Current	Post-rehabilitation
Hydrology	D	C
Geomorphology	C	C
Vegetation	E	C
Overall	D	C
Functional wetland habitat (ha)	55.44	66.22

Note: A shift from Present Ecological State class ‘D’ to ‘C’ represents an improvement in the ecological integrity of the wetland.

Source: Adapted from the Mthinzima wetland rehabilitation plan (GroundTruth, 2015).

Box 1: Description of Present Ecological State classes

Present Ecological State class	
A	Unmodified, natural.
B	Largely natural with few modifications. A slight change in ecosystem processes is discernible and a small loss of natural habitats and biota may have taken place.
C	Moderately modified. A moderate change in ecosystem processes and loss of natural habitats has taken place but the natural habitat remains predominantly intact
D	Largely modified. A large change in ecosystem processes and loss of natural habitat and biota has occurred.
E	The change in ecosystem processes and loss of natural habitat and biota is great but some remaining natural habitat features are still recognizable.
F	Modifications have reached a critical level and the ecosystem processes have been modified completely with an almost complete loss of natural habitat and biota.

Based on the condition of the wetland at the time of assessment, the approximately 98 ha wetland was considered equivalent to 55.44 ha of functional wetland habitat, based on its ecological integrity rating and size. Through the rehabilitation efforts it was projected that the ecological integrity of the wetland would improve increasing the area equivalent of functional wetland to 66.22 ha. The functional wetland area under the two scenarios is illustrated in Figure 5.2. Following this approach, the rehabilitation is expected to result in a 10.78 ha gain in functional wetland habitat. The 10.78 ha habitat gain was combined with nitrogen removal rates for restored wetlands to estimate the additional nitrogen reduction potential of the wetland with rehabilitation relative to the present state. Unit area nitrogen retention rates were sourced from the literature, specifically from the database of a systematic review of studies of the removal of nitrogen and phosphorous by restored and created wetlands (Land et al., 2016).

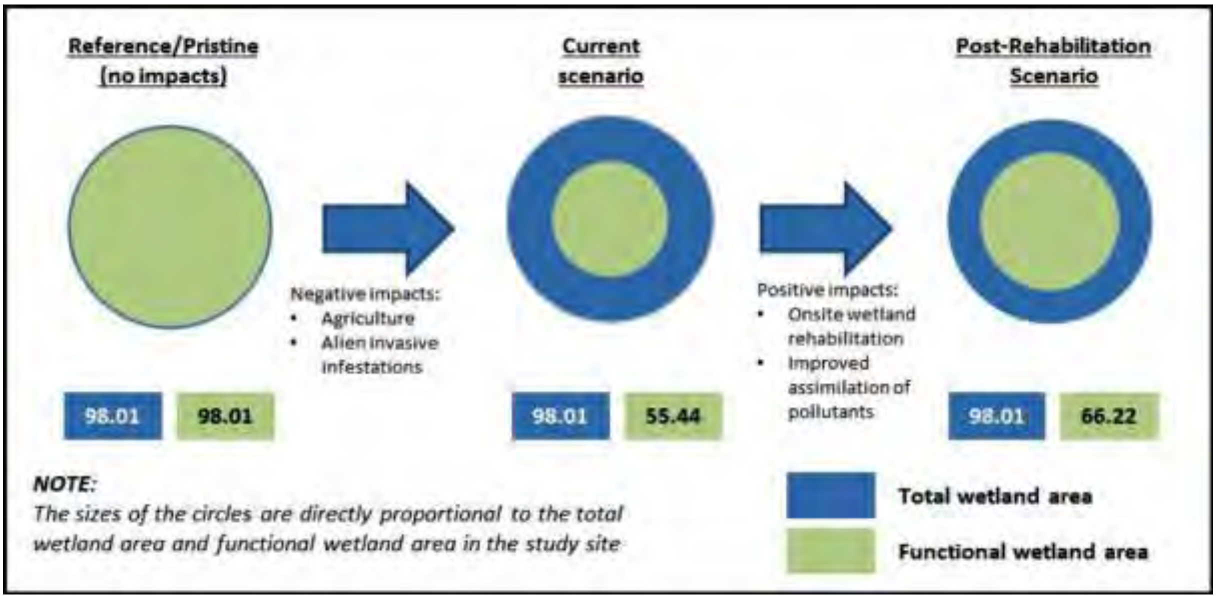


Figure 5.2: A graphic representation of the projected change in functional wetland area associated with the proposed rehabilitation of the Mthinzima wetland.
Source: Reproduced from the Mthinzima wetland rehabilitation plan (GroundTruth, 2015:21).

A number of factors affect the nitrogen removal effectiveness of a wetland and there is considerable variation in removal rates between wetlands as evident in the broad range reported by Land et al. (2016) (Table 5.2). Ideally, removal rates should be sourced from local studies of similar wetlands supported by primary data collection; however, at the time of this case study, there were no published long terms studies, or monitoring results, of nutrient removal by rehabilitated wetlands (or wetlands in general) in the region (KwaZulu-Natal). In a study of wetlands in the South Western Cape, Turpie et al. (2010) adopted an indirect approach to estimate the water treatment capacity of wetlands on a landscape scale by relating the water quality at catchment outflow points to the prevalence of wetlands within the catchment using multivariate statistical analysis. Wetlands were found to play a significant role in the reduction of nitrates, nitrites, and ammonium, but not dissolved phosphorus or suspended solids. The authors note that the nitrogen removal rate estimates (Table 5.2) were higher than expected.

Applying removal rates from the international literature introduces considerable uncertainty into the study. The database of Land et al. (2016) allows for filtering based on a number of factors affecting nitrogen removal such as wetland type and climate. The database was filtered to reflect conditions similar to the case study context, specifically: wetland type (free water surface), wetland history (restored), climate zone (Cfb which corresponds to case study area (Conradie, 2012)). While a step toward improving the reliability of the results, filtering the database in this way reduced the number of studies to only four wetlands, all located in Europe. The median nitrogen removal rate across the four studies was 69 kg of nitrogen removed annually per hectare of wetland habitat. This value was applied in the case study as a conservative estimate of the nitrogen removal rate of wetland habitat.

Table 5.2: Nitrogen removal rates for restored and created wetland systems

Source	**Removal rate (kg/ha/yr)**			**Location**
	Point estimate		Range	
Land et al. (2016)	Median, N=255	930	-3 to 12 700	Multiple - predominantly North America & Europe
Land et al. (2016) - filtered	Median, N=4	69	-3 to 337	Filtered – All Europe
Turpie et al. (2010)	Average	1 594	307 to 9 505	South Africa

5.3.3 Valuing the gain in wetland nitrogen removal potential

The economic value of nitrogen removal by a wetland is a function of the benefits of clean(er) water to the downstream system such as lower treatment costs to produce potable water and recreational use of water bodies. Hence, the direct approach to valuation is to ascertain and assign an economic value to the benefits of water entering the downstream system containing lower nitrogen loads. Where data and ecological change models are not available for estimating the service-benefit relationships required for the direct approach, the replacement cost method can be applied as an indirect approach. The replacement cost method allows for a ‘proxy’ value of wetland nitrogen removal to be estimated based on the marginal costs of nitrogen abatement achieved through an alternative means.

The replacement cost method is considered an indirect valuation approach in that the 'value estimate' does not relate specifically to willingness to pay for the benefit (or willingness to accept compensation for a benefit loss) (Barbier et al., 1997; Heal et al., 2005). Strictly, cost-based methods do not provide a measure of economic value. It is for this reason that acceptance of the method as a calculus of the economic value of ecosystem services is debated. The replacement cost reflects the cost to achieve a similar level of service through an alternative means, but gives no indication of whether people would actually pay the estimated 'replacement cost' and reflects little about the benefits of the service provided in a specific way (i.e., by an ecosystem) and the associated demand. The limitations of the replacement cost method and a number of caveats associated with its application in the context of this case study are taken up in the discussion.

Lacking the data and models required for the direct approach, the replacement cost method was applied in this case study under the assumption that a reduction in nutrient inputs to the impoundment (Midmar Dam) is preferred (by the public) over increased risk of eutrophication. In other words, it was assumed that any additional water quality enhancement potential of the wetland as a result of the rehabilitation would be fully demanded. This assumption is supported at a national level in the National Water and Sanitation Master Plan (DWS, 2018:7-72) specifically Target 1 of the High-level Water Quality Management Master Plan Targets "By 2030, water in, or from water resources shall be fit for use" which is aligned with the National Development Plan 2030 and Sustainable Development Goal 6. The plan advocates a precautionary approach[79] in the management of wastewater. Locally, a reduction in water pollution to Midmar Dam, notably nitrogen inputs associated with sewage contamination of the Mthinzima Stream, has been identified as a priority management action (WRC, 2002; GroundTruth 2012; Jogiat, 2013; Umgeni Water, 2019) particularly in light of the additional cost of treatment associated with eutrophic waters (Pretty et al., 2003; Graham et al. 2012) and the recreational value of Midmar Dam (Vundla et al., 2017). Midmar Dam is the setting for the internationally recognized Midmar Mile swimming event, the world's largest open water swimming event, which generated an

[79] The precautionary approach concept is variously defined, but generally reflects the notion that scientific certainty is not a prerequisite for taking preventive measures in the context of threats of serious or irreversible damage.

estimated direct economic impact in the region of R40 to R70 million for 2019 (Tourism KwaZulu-Natal, 2019). The uMngeni River system is also famous for the Dusi Canoe Marathon, an internationally recognized freshwater canoe paddling event. In recent years, the event has attracted much 'poor press' as a result of deteriorating water quality (see for example Pieterse, 2018) which has been associated with declining event participant numbers (Houdet et al., 2020). The public outcry regarding the declining water quality of Midmar Dam itself (e.g., The Witness, 2015; Pillay, 2019a; 2019b) and of the greater uMngeni system, specifically in the context of sport and leisure use, is an indication of the social value of clean water for recreation. Turpie et al. (2010), in a study of the water quality amelioration value of wetlands in the Western Cape, similarly assumed that any additional water quality enhancement potential was fully demanded, arguing that such an assumption was reasonable in the South African context.

The replacement cost method requires that an alternative means of achieving a reduction in nitrogen levels in the Mthinzima Stream be identified and that the cost to provide an equivalent level of reduction as that gained as a result of the rehabilitation by means of the alternative be established. Consultation with relevant experts[80] was used to explore potential alternatives to wetland nitrogen abatement. Suggestions included upgrading and extending the waterborne sewage network which would also need to address misuse of the system; installing floating wetlands into the Stream, a relatively new, experimental technology for improving water quality; and treatment by a conventional, relatively low volume wastewater treatment plant[81]. Due to a lack of information on the practical implementation, effectiveness and costs of the floating wetland alternative, and the likely significant costs associated with upgrading the entire sanitation network, conventional (built infrastructure-based) wastewater treatment was considered as the closest alternative. Conventional wastewater treatment is commonly adopted as the alternative in replacement cost valuation of wetland water quality enhancement (Chapter 3). The question of suitable alternatives and equivalent service potential is taken up further in the discussion section.

[80]The consultation included hydrologists and engineers familiar with the system and private sector providers of water treatment solutions.
[81] A 1ML/day treatment capacity was suggested.

Costs of nitrogen abatement by conventional wastewater treatment processes were collated from various local and international sources and compared (Table 5.3). Except for the study by Gren (1995), the abatement costs are based on the operating and maintenance costs of a wastewater treatment works (WWTW). Due to economies of scale, the operating and maintenance costs per unit nitrogen removed are higher for smaller treatment plants. Accordingly, the wastewater treatment cost curve estimated by Turpie et al. (2010) was used to derive the nitrogen abatement cost associated with a relatively small conventional (1ML/day) WWTW. The estimate of R144/kg nitrogen (N) reduction derived from Turpie et al. (2010) was used as the replacement cost value for wetland nitrogen removal.

Table 5.3: Nitrogen abatement costs by conventional wastewater treatment works

Source	**R/kg N-reduction**
South African sources	
Turpie et al. (2010) – reported nitrogen removal cost	
Large capacity facility	37
A 1 ML / day capacity facility	144
Naidoo et al. (2016) – derived from reported operating costs of WWTW	
Large plant (164 ML/day)	42
Small plant (0.04 ML/day)	160
International studies	
Grossman (2012) – reported marginal nitrogen removal cost	
Low estimate	39
High estimate	276
Birch et al. (2011) – reported nitrogen removal cost	
Low estimate	606
High estimate	3 230
Gren (1995) – reported nitrogen removal cost (investment & operating cost)	
Low estimate	472
High estimate	5 007

Note: All estimates adjusted to 2017 Rands. Estimates calculated from operating costs assume nitrogen removal by WWTW of 33 kg/ML (Turpie et al., 2010).

5.3.4 Data collection

Data for the case study were gathered from a number of sources including the department leading the rehabilitation project (KZN Department of Economic Development, Tourism and Environmental Affairs), the relevant water supply utility (Umgeni Water), the consultants responsible for the rehabilitation plan (GroundTruth), the engineering firm associated with the proposed Mpophomeni WWTW (Royal Haskoning DHV), researchers involved in studies of the catchment (UKZN Centre of Water Resources Research) and the literature. Data were gathered through meetings, telephonic and email consultation, expert workshops, focus group sessions, document review and expert accompanied site visits.

5.4 RESULTS

5.4.1 The estimated value of the additional wetland nitrogen removal potential

Based on the hectare equivalents approach described in section 5.3.2, implementation of the proposed rehabilitation of the Mthinzima wetland is expected to lead to a gain in functional wetland habitat of 10.78 ha. Combining the gain in functional habitat with a nitrogen removal rate of 69 kg/ha/year generates a conservative estimate of the gain in nitrogen reduction potential of 744 kg/year as a result of the rehabilitation. Applying the waste water treatment cost curve estimated by Turpie et al. (2010), a nitrogen abatement cost of R144/kg associated with a small (1ML/day) WWTW was applied to the estimated gain in nitrogen removal potential to generate a conservative 'replacement cost' value estimate of R107 332/year (Table 5.4, 2017 prices). This estimate is a proxy for the value of the annual benefit of the wetland rehabilitation in terms of enhanced nitrogen removal, assuming that the gain in removal potential can be fully utilized (i.e., the wetland is afforded the opportunity to provide the additional level of service, see Section 5.5.3), and the additional level of service is fully demanded (see Sections 5.3.3 and 5.5.6).

Table 5.4: Estimated value of the gain in wetland nitrogen removal potential with rehabilitation, 2017

Variable		**Estimate**
Gain in functional habitat as a result of rehabilitation	ha	10.78
Nitrogen removal rate	kg/ha	69
Cost of nitrogen abatement	R/kg	144
Value of gain in nitrogen removal potential	**R/year**	**107 332**

Note: All monetary values are 2017 Rands; discrepancies due to rounding.

5.4.2 Cost-benefit analysis of wetland rehabilitation from a nitrogen removal service perspective

The economic efficiency outcomes of the wetland rehabilitation were analysed within a partial cost-benefit framework by combining the estimated value of the gain in nitrogen reduction service of the wetland with rehabilitation (the benefit) with the estimated cost of the wetland rehabilitation. The analysis is necessarily incomplete, as it does not reflect the influence of the rehabilitation on additional wetland ecosystem services (e.g., erosion control and carbon storage). The investment (capital) cost of the wetland rehabilitation was budgeted at R12 million, as reflected in several funding applications submitted by the KZN Department of Economic Development, Tourism and Environmental Affairs (Felton, 2017). An annual maintenance cost of R300 000 was applied based on de Groot et al. (2013) who allowed for maintenance costs of 2.5% of investment cost for wetland rehabilitation.

As the base case, a 40 year lifespan was applied; a 40 to 50 year basic life is considered plausible for wetlands designed for water treatment (Kadlec and Wallace, 2009). There is on-going debate over an appropriate discount rate for ecosystem valuations; there are no purely economic guidelines for choosing a discount rate and the choice is really an ethical one. Frequently, in ecosystem valuations, the usual discount rate for public investments for the relevant country is applied. This approach is adopted as the base case and a real discount rate of 8% is applied following Mullins et al. (2014). However, both the notion of 'discounting' and the rate at which future benefits - especially those that flow from natural capital - should be discounted, are a contentious topic within environmental valuation and lower discount rates are often argued in the context of ecosystem service valuation (Fenichel et al., 2017). In this case study, a discount rate of 3% was applied as a comparison.

The results, Table 5.5, indicate that the annual benefit measured as the value of the gain in nitrogen removal by the wetland with rehabilitation, derived from the operating costs of conventional waste water treatment, is insufficient to generate a positive net welfare effect across timeframes of 40 and 20 years and at discount rates of both 8% and 3%. Arguably, basing the benefit valuation solely on the operating costs of a WWTW is a poor reflection the 'replacement cost' in this case, as the alternative nitrogen abatement option could not be achieved without first constructing a standard, low volume wastewater treatment plant. Gren (1995) included capital expenditure in the replacement cost estimate for a study of the Stockholm archipelago; similarly, Russell and Greening (2015) included lifecycle costs associated with building wastewater treatment facilities as an upper limit wetland replacement value. Grossman (2012) also considered capital expenditure in the form of construction and upgrades to WWTW in estimating a shadow price of floodplain nutrient retention in the Elbe River Catchment.

Table 5.5 shows the results of an additional analysis incorporating the estimated construction costs of a standard low volume wastewater treatment plant[82] in the benefit value estimate. For comparison, an analysis was also conducted using the average of the low and high nitrogen abatement costs reported by Gren (1995) (Table 5.3) based on the combined operating and investment costs of conventional wastewater treatment.

Table 5.5: Results of the partial cost benefit analysis showing the effects of different benefit estimate approaches, discount rates and timeframes on the benefit-cost ratio

Benefit estimate approach		8% discount rate		3% discount rate	
		20 year	40 year	20 year	40 year
i)	Replacement cost of gain in annual nitrogen removal	0.07	0.08	0.10	0.13
ii)	Combined capital (construction cost) and (i)	1.05	1.02	0.98	0.90
iii)	Benefit estimate based on nitrogen removal cost reported by Gren (1995) incorporating capital cost	1.34	1.56	1.84	2.49

An economic efficiency gain is achieved when the estimated construction costs of a standard low volume WWTW are incorporated in the benefit estimate at an 8% discount rate (approach ii in Table 5.5). Lowering the discount rate to 3% in this case, reduces the benefit-cost ratio as less 'weighting' is placed on the initial benefit (WWTW construction cost avoided) which exceeds the upfront wetland rehabilitation cost. This result emphasises the sensitivity of the analysis to the choice of discount rate.

Under the second benefit estimate approach (approach ii in Table 5.5), the benefit-cost outcome is sensitive to the inclusion of the annual wetland maintenance costs as shown in Table 5.6. Excluding maintenance costs increases the benefit-cost ratio across both discount rates and timeframes. The effect is more pronounced under a 3% discount rate and a longer timeframe as both these choices favour longer term ongoing benefits and costs compared to higher discount rates and shorter timeframes which place greater weight on earlier benefits

[82] Construction cost sourced from Turpie et al. (2010); the CPI adjusted value (R 14.6 million) compared favourably to a cost estimated provided by a local engineering firm experienced in WWTW construction (R15 million) as well as to quotes sourced from other local suppliers of conventional wastewater treatment plants.

and costs. Under a 3% discount rate, excluding maintenance costs from the analysis changes the outcome from a negative net benefit of rehabilitation to a positive net benefit outcome.

Table 5.6: Sensitivity analysis results of the effect of wetland maintenance costs on the benefit-cost ratio

Benefit estimate approach: ii) Combined capital (construction cost) and annual nitrogen removal	**8% discount rate**		**3% discount rate**	
	20 year	40 year	20 year	40 year
Rehabilitation cost inclusive of annual wetland maintenance costs	1.05	1.02	0.98	0.90
Rehabilitation cost exclusive of annual wetland maintenance costs	1.30	1.32	1.35	1.42
Difference	*0.25*	*0.30*	*0.37*	*0.52*

Ongoing maintenance is necessary to ensure the long-term success of the rehabilitation intervention and includes activities such as monitoring the integrity of rehabilitation structures, minor repairs to structures and alien plant and wetland biomass management. Iftekhar et al. (2017) note that maintenance costs are often left out of estimates of ecological restoration projects. The wetland restoration studies reviewed in this thesis (Chapter 3) similarly indicate that ongoing maintenance costs are not always included in cost estimates. On the other hand, it could be argued that this maintenance cost is not solely attributable to the rehabilitation, and that part of the cost would be incurred to maintain the wetland in an acceptable condition even if it had not first required rehabilitation (e.g., alien plant and wetland biomass management).

In comparison, adopting the nitrogen abatement value reported by Gren (1995) (approach iii in Table 5.5) produces an economic efficiency gain across all discount rates and timeframes. In the case of applying the Gren (1995) nitrogen abatement value, the annual benefit of additional nitrogen reduction now exceeds the annual maintenance cost associated with the wetland rehabilitation. The comparison illustrates the variability in evaluation outcomes depending on the 'replacement cost' value selected to reflect the benefit value. The broad range in nitrogen abatement costs from the literature presented in Table 5.3 (a range of R37 to R5 007 per kg nitrogen removed) emphasises how variable the cost-benefit evaluation outcomes could be depending on which abatement cost was applied.

While commonly applied as an indirect benefit valuation approach, the replacement cost method does not strictly reflect the value of the rehabilitation benefit (see sections 5.3.3 and 5.5.7). As such, the method, and how it has been applied in this case study, could be framed as a cost-effectiveness analysis rather than a partial cost-benefit analysis. Such a framing would still provide valuable information to decision makers, without making any judgments on the benefits of the wetland rehabilitation or suggesting that the only value of the rehabilitation is the saving in water treatment costs.

5.5 DISCUSSION

The estimated economic value and corresponding CBA indicates that an economic efficiency gain is achieved when the projected costs of the construction and operation of a standard low volume WWTW are used as the replacement value for the rehabilitation. However, several fundamental assumptions and caveats warrant further consideration in interpreting the study results; the simple cost-benefit decision making rule should not be applied in this case without careful consideration of these aspects (discussed in detail below).

From a reflection on this case study application and the associated assumptions and challenges, a set of eight factors to consider when using the replacement cost method in wetland rehabilitation valuation and decision-making have been derived (Table 5.7). Careful consideration of these aspects in applying the replacement cost method can improve the reliability of the results and inform the interpretation of, and level of confidence in, the results for use in wetland rehabilitation evaluation and decision-making.

Table 5.7: Factors to consider when using the replacement cost method in wetland rehabilitation evaluation and decision-making

Enhances reliability / usefulness	Reduces reliability / usefulness	Case study rating of reliability
1) The biophysical response of the wetland to the rehabilitation is assessed. The 'with' and 'without' rehabilitation scenarios can be formulated based on established science and the *difference* in biophysical function and associated ecosystem service flows between them quantified through primary data collection or available site-specific data. There is a detailed plan for, or record of, the rehabilitation works.	The scenarios can only be described conceptually due to limitations in scientific understanding and or availability of context specific information; only secondary data, or estimates derived from the literature are available, especially if the context of the data source(s) differs markedly to the study site or the differences cannot be ascertained.	Low
2) The biophysical differences reflect non-critical changes in wetland 'stock' and ecosystem service flows that do not fundamentally affect the value of an additional unit of service.	There is high uncertainty regarding critical threshold levels or concern that the assessment site is approaching a threshold level; the benefit value / replacement cost is particularly sensitive to even small changes in the biophysical context. In this case, the implications for the economic value estimate of critical changes in the system should be considered (i.e., whether the marginal replacement cost value remains appropriate).	Moderate
3) There is demonstrated opportunity / ability to derive benefit from the change in ecosystem service flows.	There is limited opportunity to derive benefit from the change in ecosystem service flows or there are barriers to access additional ecosystem service flows.	High
4) The priority ecosystem services / benefits – those relevant to the specific objectives of the rehabilitation - are amenable to the replacement cost method, typically the provision of material resources, streamflow regulation and water quality enhancement, and the option to apply other more direct valuation methods is constrained.	The priority ecosystem services / benefits for valuation are not amenable to the replacement cost method (this links to the point below on the presence of a realistic replacement alternative).	Moderate
5) There is a realistic replacement (alternative) that can provide the same service, and it is the least-cost alternative. In effect, the replacement cost approach assumes substitutability between alternatives in achieving the intended outcome. A judgement on 'realistic' alternatives and whether they provide the same service will be influenced by a 'strong' vs. 'weak' sustainability position.	There is no realistic alternative or the service / benefit provided by the alternative(s) differs from that provided by the wetland rehabilitation and the 'replacement cost' cannot be isolated for the specific service / benefit. There is limited opportunity for a rigorous context specific investigation of the outcomes of various possible alternatives and a corresponding cost analysis.	Low

Enhances reliability / usefulness	Reduces reliability / usefulness	Case study rating of reliability
6) It is possible to derive the cost of the alternative from actual expenditure or it can be confidently approximated for the local context. All relevant cost components (e.g., OPEX and CAPEX) costs are included / considered.	It is not conclusive that the replacement cost estimate accurately reflects local conditions. There is uncertainty regarding the components that should be included in the replacement cost estimate.	Moderate
7) There is substantial evidence demonstrating a demand for the benefits of the additional ecosystem service flows and that the equivalent level of service / benefit would be demanded if it was provided by the replacement alternative.	There is little information available to inform a judgement on the demand for the additional ecosystem service flows; it appears that people are not willing to pay to replace lost benefits as a result of wetland degradation or are unlikely to actually pay the estimated 'replacement cost'.	High
8) The replacement cost method is recognized as a 'partial analysis' (targeted at the primary objectives of the rehabilitation) and the assessed benefits are placed within a context of other likely outcomes (positive and negative) and potential values. The distribution of potential 'benefits' and 'dis-benefits' across differ groups is considered.	Consideration / description of the broader benefit, dis-benefit, value context and their distribution across different groups is limited.	Moderate

5.5.1 Assessment of biophysical response - nitrogen removal uncertainty

The economic valuation of wetland rehabilitation is necessarily specific to the value of the outcomes as a direct result of the rehabilitation efforts rather than the value of the entire (restored) wetland (Bockstael et al., 2000; Turner et al., 2003; Polasky & Segerson, 2009; Keeler et al., 2012). These outcomes pertain to the *difference* in biophysical function and associated flow of ecosystem services between the 'with' and 'without' rehabilitation scenarios.

For reliable evaluation of the economic value of wetland, rehabilitation, the biophysical response of the wetland to the rehabilitation must be assessed as indicated by Factor 1, Table 5.7. This form of assessment is loosely referred to as 'marginal valuation' in contrast to 'total valuation' which considers the value of the whole (restored) ecosystem rather than the value of a specific change in the ecosystem (Ricketts and Lonsdorf, 2013).

In this case study, the expected gain in nitrogen retention potential of the wetland as a result of the proposed rehabilitation was estimated. It was to this 'gain' that an economic value was assigned. The estimation was informed by various ecological assessments comparing a projected 'with rehabilitation' state to the present wetland state and the corresponding difference in area equivalent of functional wetland habitat between the two states. In this way, the economic valuation was related directly to the biophysical influence of the rehabilitation interventions and the resulting outcomes.

A primary assumption of the case study is that the functional wetland habitat of the Mthinzima wetland has the potential to retain nitrogen at a rate similar to that observed in other cases of wetland rehabilitation. This assumption is supported by sporadic water quality monitoring upstream and downstream of the wetland (van Deventer, 2012; Terry, 2017) and ecological health and ecosystem service assessments of the wetland (GroundTruth, 2015). However, the actual nitrogen retention by the wetland was not measured directly. Nitrogen removal rates available from the literature were adopted and applied across the gain in functional wetland habitat. As noted by Turner et al. (2003:499), the "technique of 'benefits transfer' is, however, fraught with difficulties". This case study highlights a number of such difficulties in the case of wetland nitrogen removal.

Uncertainty associated with the **nitrogen removal rate** applied in this case study stems from whether the rate drawn from the literature is an accurate representation of the nitrogen retention capacity of the Mthinzima wetland – that is, uncertainty regarding the reliability of transferring retention rates from the literature to the study site – and to applying a constant rate of nitrogen retention both spatially (across the gain in wetland habitat) and temporally. Nitrogen retention by wetlands is a result of several independent processes; each influenced by a number of factors (Marton et al., 2015). Variation within these site-specific factors leads to considerable variation in nitrogen removal rates across wetlands. The meta-analysis of Land et al. (2016) reported a range in nitrogen removal rates of -3 to 12 700 kg/ha/year and found retention rates to depend significantly on inlet nitrogen loading rate, wetland area (size) and annual average air temperature.

Nitrogen removal rate and **wetland size** appear to be negatively correlated (Land et al., 2016). The nitrogen removal rate applied in this case study, drawn from a sub-set of studies from the Land et al., (2016) database, is associated with an average wetland size of 473 ha. Given the total functional area of the Mthinzima wetland of 66 ha, the applied nitrogen removal rate is possibly an underestimate. However, the size of the wetlands evaluated by Land et al. (2016) ranged from 0.0001 ha to 1000 ha and the rate of decline in removal rate with wetland size appeared to be lower for areas greater than 1 ha.

Air temperature appears to have a positive influence on nitrogen removal rate (Land et al., 2016). While the database of Land et al. (2016) was filtered according to climate zone Cfb (warm, fully humid, warm summer) which corresponds to the case study area (Conradie, 2012), the average air temperature across the sub-set of studies was 8.7^{o}C. This is lower than the average air temperature of the ecoregion of this case study which is 15°C (WRC, 2002); again, suggesting that the rate applied in this assessment is conservative.

The inlet nitrogen loading rate is a key factor influencing the rate of nitrogen retention by the wetland. Nitrogen removal rate has been shown to be positively correlated to inflow nitrogen concentration, with a steeper increase in removal rate at concentrations greater than 18 mg/L, and with greater hydraulic loading, up to a point (Land et al., 2016). Insufficient timeseries data were available to establish the flow and nitrogen concentration for the inflow to the Mthinzima wetland over time. Sampling results for a site upstream of the wetland and downstream of Mpophomeni in 2010-2011 indicated an average nitrogen concentration of 8.9 mg/L with a range of 1 mg/L to 18.8 mg/L (5 samples) (van Deventer, 2012). Inflow nitrogen concentrations were not reported for the sub-set of studies from the Land et al. (2016) database and no comparison could be made in this regard. Average inflow nitrogen concentration across all the studies evaluated was 15.7 mg/L (1.3 to 115.2 mg/L), suggesting that the inflow concentrations to the Mthinzima wetland are within the range evaluated in the meta-analysis of Land et al. (2016).

As a result of the lack of timeseries data on nitrogen concentration and stream flow / hydraulic loading, a constant nitrogen removal rate for the wetland was assumed in this case study. The variation in nitrogen concentrations across the sampling by van Deventer (2012) highlights that, in the case of the Mthinzima wetland, nitrogen inputs to the wetland system are not constant, suggesting that nitrogen removal rates will also be dynamic. Water quality and nutrient loads within the catchment are event driven influenced by rainfall and the resulting runoff and pollution incidents (e.g., surcharging sewer). Seasonal variance in precipitation is partially taken into account in the nitrogen removal rate drawn from Land et al. (2016) as the rate estimates reflect the average removal over an annual cycle and the wetlands of the sub-set of studies from which the removal rate was drawn were also precipitation driven. The mean annual precipitation for the study ecoregion is 930 mm (WRC, 2002); whereas the average for the sub-set of studies from the Land et al. (2016) database was 698 mm. However, effects due to the nature of the precipitation (e.g., storms versus light rainfall) are not accounted for in applying nitrogen removal estimates from the literature.

The significant influence of inlet nitrogen loading on wetland nitrogen removal rate adds to the uncertainty associated with drawing from the literature for removal rates. This uncertainty may be partly addressed if it can be established that the inflow loading and primary nitrogen sources (e.g., agricultural runoff or secondary wastewater) of the study site and the studies from the literature are similar, or if there is sufficient information available for both cases to allow the nitrogen removal rate to be adjusted to account for key differences between the sites. However, potential variation in nitrogen removal rates associated with variation in inlet loading rate due to site specific climate and pollution events is not straightforward to address. Nitrogen concentrations in the Mthinzima Stream are variable, as demonstrated by van Deventer (2012); accordingly, it can be expected that nitrogen removal rates would be dynamic over time. Patterns of temporal variation in nitrogen removal rate will be different across wetlands depending on local events (e.g., climatic and pollution events) and, without primary timeseries data, the extent of this temporal variation and to what degree it affects average removal rates, is not clear. Average rates determined from long-term monitoring data (greater than a single annual cycle) would help to reduce the uncertainty.

The positive relationship between inflow nitrogen loading and wetland nitrogen removal rate also has implications for the nitrogen removal rate applied under different scenarios. For instance, in this case study, in assessing the value of wetland rehabilitation in the case of the implementation of a new WWTW upstream of the wetland, it may be that the nitrogen removal rate should be adjusted downwards as the incoming nitrogen load would be expected to be lower. This suggests that any scenario analysis in which wetland inflow nitrogen load is affected should consider adjusting the wetland nitrogen removal rate applied. While conceptually appealing, the empirical studies and general relationships on which to base such adjustments are not yet commonly available in the South African wetland context.

Further to temporal variation, biogeochemical cycling of nitrogen varies spatially within and across wetlands (Marton et al., 2015). By assuming a constant nitrogen retention rate per hectare for the gain in area of wetland with the rehabilitation, this case study has not explored the potential heterogeneity in nitrogen retention within the wetland. To a degree, potential spatial variance is partially addressed by applying the 'functional hectare equivalent' approach to estimate the gain in wetland habitat. The 'functional hectare equivalent' approach suggests that each hectare of wetland gain included in the analysis is 'functional' and theoretically able to provide a similar level of ecosystem services. While an improvement on a simple 'area of wetland rehabilitated' approach, each 'functional' unit of wetland habitat may not have the same potential to retain nitrogen, given hydrologic flow paths, differences in plants species and distributions and variability in soil properties (Kotze et al., 2008; Marton et al., 2015). The nitrogen removal rate applied in this case study was estimated from nitrogen input-output studies of wetlands multiple hectares in size and it can be expected that the measured rates were a result of some level of intra-wetland heterogeneity. However, how the intra-wetland heterogeneity of the primary wetland sites and this case study site compare could not be ascertained from the database of Land et al. (2016) beyond the classification of wetland type. Furthermore, it is not yet clear the causal processes and influencing factors behind spatial variability in nitrogen cycling processes within wetlands (Marton et al., 2015).

A further caveat in applying nitrogen retention rates drawn from the meta-analysis by Land et al. (2016) relates to geographical (predominantly northern hemisphere studies with no cases from Africa) and potential publication bias[83] in the results as well as the small sample size after filtering the Land et al. (2016) database to align with the case study wetland settings (n = 4). The inherent uncertainty in drawing nitrogen removal rates from the literature, particularly from a base of literature geographically distant from the case study and given the numerous site-specific factors that affect the nitrogen retention of a wetland as well as spatial and temporal variation in retention, highlights the need for studies of wetland nitrogen dynamics in South Africa supported by water quality and flow monitoring. Such studies should aim to consider, among other aspects, nitrogen species and their interactions, temporal and spatial resolution, event impacts and scenario analyses of alternative upstream and wetland transition settings.

In the context of rehabilitation, an additional aspect requiring consideration is the response time or 'lag' between implementation of the rehabilitation and the response of the wetland. It is unlikely that the additional nitrogen retention capacity would be gained in the short-term (within a year) (GroundTruth, 2015), but would increase over the first few years following the rehabilitation as the new vegetation and flow paths became established. The results of local empirical studies of wetland nitrogen removal (and additional water quality enhancement capacities) and the influence of rehabilitation interventions on removal potential would improve the accuracy of the valuation of wetland rehabilitation from a nitrogen removal

[83] Publication bias stems from the potential under-reporting of findings of low nitrogen retention rates or even net releases of nitrogen from the wetland, although Land et al. (2016) consider this likely to be insignificant.

perspective. Particularly for event driven or varying nitrogen loading, comprehensive monitoring at regular intervals is needed.

The biophysical response assessment was based on a comparison of a projected ‘with rehabilitation’ state to the present wetland state, which does not consider the likelihood of continued degradation of the wetland without rehabilitation. The estimated economic value of the rehabilitation is based on a counterfactual scenario assuming that without rehabilitation the wetland would remain in its current state – degraded, but still providing some level of water treatment service. In reality, it is likely that without rehabilitation the wetland would continue to degrade resulting in a further loss in wetland services (Terry, 2017). As such, the long-term value of the wetland rehabilitation is understated. Additional scenarios of possible states of the wetland overtime would increase the depth of the analysis and understanding of the value of the rehabilitation and enhance the assessment as a decision-aid. Any changes in the extent of rehabilitation efforts required, and the associated changes in rehabilitation cost, would also need to be considered.

The specific attention given to ascertaining the difference in nitrogen reduction potential as a result of the rehabilitation strengthens the confidence in the assessment as a reflection of the value of the rehabilitation. The lack of nitrogen load data and reliance on the literature for a nitrogen reduction rate due to the paucity of local empirical studies of wetland nitrogen dynamics, reduces the overall confidence in the estimates. As a result, Factor 1 (Table 5.7) has been assigned a ‘low confidence’ rating for this case study, suggesting that the results are indicative rather than definitive; the estimated gain in nitrogen retention is likely understated.

5.5.2 Valuation at the margin

By definition, economic value is measured at the ‘margin’ reflecting the utility of an additional unit of ecosystem service assuming non-critical changes in the underlying ecosystem stock (Bateman et al., 2011). For economic value estimates to be robust, the magnitude of the biophysical changes / difference must fall within a non-critical range that does not fundamentally affect the value of an additional unit of service (Turner et al., 2003; Farley, 2012). This aspect of economic valuation is reflected as Factor 2 in Table 5.7.

In this case, the expected outcomes of the rehabilitation efforts are associated with relatively small-scale changes both at the level of the wetland (a 19% change in functional habitat and a corresponding 19% gain in nitrogen retention service) and relative to the scale of reduction in load of nitrogen to the Midmar Dam. The estimated increase in nitrogen retention capacity of the wetland with the rehabilitation under the conservative scenario is equivalent to 3.2% of the surface inflow of nitrogen to the Dam from its major rivers as determined by Namugize (2017)[84] suggesting that rehabilitation of the wetland alone will not reduce all risk of eutrophication. Similarly, it is not definitive that the case without rehabilitation (under status quo conditions) would, alone, result in eutrophication of the Dam (i.e., the reaching of a threshold level rather than a marginal change in eutrophication risk). From this consideration of the scale of change, it can be assumed that the effects of the rehabilitation are not of an order of magnitude to influence the marginal value of nitrogen abatement within the system.

However, this is a much-simplified analysis and interpretation. Evaluating environmental changes in respect of critical thresholds is complex and predicting what is a 'non-marginal' change can be far from straightforward (Barbier, 2007, Farley, 2012; Jax, 2014). In this case, the second key factor influencing the reliability of replacement cost valuation (Factor 2, Table 5.7) has been assigned a 'moderate confidence' rating as several sources of uncertainty regarding threshold levels remain.

In the context of eutrophication, neither the relationship between nutrient inputs and eutrophication level, nor the relationship between eutrophication level and water use impacts, are linear. Eutrophication incidents are event driven, following spikes in nutrient loads (Graham et al., 2012); the ecological and economic consequences of eutrophication are most often experienced as a result of changes in the level of eutrophication and the variability thereof at the time of the incident. Without limnological studies of the Dam and the modelling of various scenarios, it is not clear where in a transition path to eutrophication the Dam currently is, and the influence of nitrogen loads from the Mthinzima Stream on this 'state'. What appears here as a relatively small difference in nitrogen inputs to the Dam between the

[84] Sampling for the study by Namugize (2017) was carried out during severe drought conditions, and the nutrient loads may be underestimated. Atmospheric deposition is an additional source of nitrogen input to the Dam further increasing the total nitrogen loading to the Dam.

‘with’ and ‘without’ wetland rehabilitation cases (3.2% of the surface inflow of nitrogen to the Dam), may in effect be close to, or influence, a eutrophication threshold level. Adherence to the condition of ‘valuation under non-critical change’ can be difficult to establish and, furthermore, is influenced by the geographical and temporal scale of the assessment.

Nitrogen abatement is a local (to regional) ecosystem service, the utility of which might be expected to increase as the risk of eutrophication increases or the system nears a eutrophication threshold. On the other hand, the marginal value of nitrogen abatement might be expected to decline with a relatively large-scale reduction in nitrogen in the system, for example through the rehabilitation of multiple wetlands or the implementation of multiple nitrogen input control interventions across the catchment. As the total input of nitrogen to the Dam is reduced, the value of each additional unit of nitrogen abatement is expected to decline. The nature of these relationship is not expected to be linear, as the relationship between nitrogen input and the risk of eutrophication is non-linear (Graham et al., 2012; Thornton et al., 2013). Limnological studies of the impoundment to improve our understanding of the internal biological and chemical functioning of the system and the relationship between changes in nitrogen inputs and eutrophication risk would better inform a judgement regarding non-critical ecosystem change.

In this case study context, rapid urbanisation of the area consisting of both formal and informal development; insecure land tenure, governance issues and contested land uses; and ongoing issues and delays in developing wastewater infrastructure upstream of the wetland, allude to much greater complexity in, and scenarios of, future wetland scenarios. In addition, factors at a regional, national and global scale (e.g., regional/national water quality objectives, increasing water insecurity at a regional/national scale) could influence the marginal value of nitrogen abatement, and, therefore, the value of the rehabilitation.

5.5.3 The opportunity for additional nitrogen reduction

Wetland rehabilitation can influence the biophysical potential of the wetland to remove nitrogen from inflows. Enhancing the capacity of an ecosystem to supply a service does not necessarily mean that the service will be realized or necessarily result in an increase in benefits for people (Wieland et al., 2016). Whether the additional biophysical potential is mobilized as a 'service' depends on the ability or opportunity to derive benefit from the biophysical function (i.e., people can 'access' the service potential[85]) and the presence of demand for the corresponding service or benefit (Spangenberg et al., 2014a). For wetland water quality enhancement, the biophysical potential of a wetland to reduce pollutants in streamflow is only mobilized in the case that the wetland receives polluted water; it is 'afforded the opportunity' to provide the service from which the benefits of improved water quality can be derived.

The economic valuation of the rehabilitation requires explicit consideration of both the access to, and demand for, any change in service supply potential because of the rehabilitation efforts. Without access and demand, an increase in ecosystem service potential has no economic value. These aspects of reliable economic valuation are reflected as Factor 3 and Factor 7 in Table 5.7. Demand in the context of this case study is discussed under section 5.5.6, this section turns to a consideration of the opportunity for the wetland to provide the additional nitrogen removal potential as a result of the rehabilitation. The rehabilitation of the wetland could result in an additional nitrogen reduction potential of 744 kg/year. The opportunity afforded the wetland to provide this additional reduction service was explored using the results from a study of nutrient loads in the upper uMngeni Catchment (1987-2013) by Ngubane (2016). Based on the simulated results, the estimated load of nitrogen from the Mthinzima Stream to Midmar Dam exceeds this capacity[86]; as such the additional nitrogen removal potential of the wetland with rehabilitation could be fully mobilized as shown in Table 5.8.

Considering the scenario of a new WWTW upstream of the Mthinzima wetland, it is expected that nitrogen loads in the Mthinzima Stream will be reduced since a WWTW would partly address the sewage management issues within the system. As described in section 5.2 and Figure 5.2, there are two main sources of sewage contamination to the Mthinzima Stream, natural seepage and overflows from the waste water storage ponds of the existing waste

[85] Wieland et al. (2016) provide a discussion of the spatial, legal, policy and economic aspects affecting 'access' to ecosystem service potential.
[86] These estimates are for a point below the existing degraded wetland; as such it is taken that the nitrogen load estimates to Midmar Dam are for the load remaining after the nitrogen retention service provided by the degraded wetland.

water pump station and run-off from failures of the waste water reticulation system. The planned WWTW will address the first source (point-source), but is unlikely to fully resolve the second. Under a scenario with an operational WWTW upstream of the wetland, the load of nitrogen to the wetland is expected to be significantly lower. As an illustration of the concept, historical nitrogen loads for a time when the Mpophomeni WWTW was operational (Ngubane, 2016) were applied (Table 5.8). The historical load of nitrogen from the Mthinzima Stream to Midmar Dam (kg/year) still exceeds the additional capacity associated with the wetland rehabilitation; as such the additional nitrogen removal potential could be fully mobilized.

Table 5.8: Opportunity for additional nitrogen reduction

		'Current load'	With WWTW load	Hypothetical load
Mthinzima Stream load to Midmar Dam [a]	kg/year	11 000	4 000	375
Additional removal capacity with rehabilitation (at 69 kg/ha/year)	kg/year	744	744	744
Remaining load	kg/year	10 256	3 256	-
Extent of potential mobilized		Additional potential can be fully realized	Additional potential can be fully realized	Additional potential can be partly realized

Note: [a] Load estimates sourced from Ngubane (2016) based on simulated flow data and limited concentration date. 'Current load' scenario based on average estimated load for the period 2012-2013. As an indication of nitrogen loads in the case of the new WWTW, historical loads for a time when the old WWTW was operational were applied (the average nitrogen load, 1996 – 1999).

While the actual estimates from this analysis are intended as illustrative, the conclusion that the gain in nitrogen removal potential of the wetland with rehabilitation could be fully realized is considered reasonable given the on-going upstream water pollution issues and the likelihood of further population expansion. With the implementation of a new WWTW upstream of the wetland, nitrogen inputs to the wetland will be reduced; however, the WWTW won't address diffuse surface run-off and pollution as a result of surcharging manholes. As such, the third aspect of reliable replacement cost valuation (Factor 3, Table 5.7) has been assigned a 'high confidence' rating in the context of this case study and the interrogation of the opportunity to derive benefit from the anticipated change in ecosystem service flows. Input-output modelling of nitrogen loads supported by water quality monitoring (nitrogen concentration and water flows) of the system would further inform this judgment.

Continuing the illustration, if the nitrogen load to Midmar Dam from the Mthinzima Stream was lower than the additional removal capacity of the wetland with rehabilitation, then only the equivalent level of the additional nitrogen retention potential could be mobilized. Hypothetically, for example, if the inflow load to the wetland was 4200 kg/year, only 375 kg/year of the additional retention potential gained through the rehabilitation could be realized (as the degraded wetland is able to provide a retention service of 3 825 kg/year). This would change the annual nitrogen retention benefit of the rehabilitation from R107 332 (full potential realized) to R54 000 (part potential realized).

This exercise illustrates the difference between an ecosystem service potential (the biophysical capacity of the ecosystem to provide a service) and the extent to which the service can be realized and emphasises that the use value (discussed below) of the rehabilitation, associated with the nitrogen retention service, is related to the degree to which the biophysical potential can be realized as well as the demand for the service or resulting benefit (i.e., the situation affords the wetland the opportunity to provide the service). The exercise is illustrative in that the load estimates applied for the WWTW scenario are not reflective of the present context and do not consider population growth, the further aging of the wastewater conveyance system, nor the larger size and specific design elements of the planned WWTW relative to the historical case. This type of assessment would benefit from input-output modelling, supported by primary measurements, of nitrogen loads (nitrogen concentration and water flows) in the Mthinzima stream and wetland system under different scenarios. Such analyses are complicated by the different forms (species) of nitrogen and the inter-conversion from one to another (e.g., ammonia to nitrite to nitrate) (DWAF, 1996) and points to the site-specific nature and complexity of water quality modelling and projections and the risks of applying simple relationships and / or estimates taken from the literature.

A caveat pertains to the interpretation of ‘opportunity’ as it applies to the present ‘use value’ of the wetland; the benefit derived through ‘use’ of that level of ecosystem service potential which can be realized at the present time. The interpretation fails to reflect option and non-use values which may apply to the ‘full’ ecosystem service potential. Option value is associated with the satisfaction derived from ensuring that a wetland service is available for future use (Barbier et al., 1997)[87] and has been likened to insurance for future possible demand (Dixon and Pagiola, 1998). Given the dynamic context of the site area – rapid urbanisation, both formal and informal, contested land-use and likelihood of ongoing failures of the sanitation system – 'option value' is an important consideration missing from this analysis.

5.5.4 The replacement alternative – equivalent, least-cost and realistic

The intention of the proposed wetland rehabilitation is to secure and improve the functioning of the system specifically to enhance the water quality services of the wetland (GroundTruth, 2015). The replacement cost method is commonly applied in valuing wetland ecosystem services related to water quality improvements (Woodward and Wui, 2001; Chapter 3). Consideration of the objectives of the rehabilitation and the corresponding priority ecosystem services / benefits (i.e., those relevant to the specific objectives of the rehabilitation and likely to be influenced by the rehabilitation efforts) and whether they are suited to replacement cost valuation is reflected in Factor 4 (Table 5.7).

For reliable replacement cost valuation, Shabman and Batie (1978) specified that the specific alternative taken as the replacement should provide the 'same service' as that of the ecosystem and be the least cost alternative, combined as Factor 5 in Table 5.7. As noted by others (Heal et al., 2005; Meyerhoff and Dehnhardt, 2007), it can be difficult to fulfil these conditions and it is unlikely in many cases that the alternatives will provide the 'same' service. What qualifies as the 'same service' is open to debate, as argued in the discussion below, and depends on the sustainability position[88] guiding the assessment which influences what may be considered as an alternative to wetland rehabilitation.

In this context, the issue of alternatives in providing a nitrogen abatement service can be approached from different perspectives. One is to consider measures for removing nitrogen

---

[87] It is contested as to whether option value is a separate component of Total Economic Value or part of 'use value' (Hanley and Barbier, 2009; Pascual et al., 2010).

[88] The strong sustainability approach views the relationship between forms of capital, specifically manufactured and natural capital, as one of complementarity rather than (aggregate) substitutability (Costanza and Daly, 1992; Farley, 2012). The weak sustainability approach assumes that different forms of capital are essentially substitutable in providing welfare, particularly manufactured and natural capital (Gutes, 1996; Dietz and Neumayer, 2007).

from the stream (a treatment approach). The second is to consider measures to reduce the input of nitrogen into the stream (a prevention approach) which entails identifying the sources of nitrogen inputs and the associated causes or drivers of pollution. A third option is to consider various measures in combination.

Commonly, replacement cost valuation of wetland water quality amelioration services is approached from a treatment perspective using the cost of nitrogen removal by conventional wastewater treatment plant as the alternative (Woodward and Wui, 2001; Chapter 3). During the conventional wastewater treatment process, nitrogen is removed from influent water. Taking a simple view, since both a wastewater treatment facility and a wetland can provide a nitrogen abatement service, they can be considered alternatives, or substitutes, in providing the service. However, several divergences between these two types of infrastructure challenge whether this may be considered as the ‘same service’.

There are both point source and non-point source nitrogen inputs to the Mthinzima Stream. The system diagram, Figure 5.2, highlights the role of the Mthinzima wetland in assimilating pollution from non-point sources (e.g., catchment run-off). Wetlands, along with other types of aquatic ecosystems (e.g., riparian vegetation), are one of the few ways to simultaneously address point and non-point source pollution. Controlling non-point source pollution is often more complex and challenging than dealing with point source discharges with few technological solutions (Verhoeven et al., 2006). In this case, a WWTW is not a practical alternative as it would require diverting the Stream through a treatment facility. Wetland systems may be particularly valuable in addressing non-point source pollution, yet this benefit is not reflected in the replacement cost of nitrogen removal by WWTW. While the two alternatives may both provide a nitrogen abatement service; they are not equivalent in their ability to address different sources of nitrogen pollution. In this way, the alternative considered as the ‘replacement’, in this case a conventional wastewater treatment plant, does not strictly provide the ‘same service’ as that of the wetland. If standard wastewater treatment remains the best available indicator of ‘replacement value’, then, arguably, additional costs reflective of stream diversion infrastructure and impacts to the aquatic ecosystem of such a diversion should be factored into the replacement cost estimate.

In practice, additional measures would be needed to address the non-point source nitrogen inputs; such as a social programme to address misuse of the sewer system and storm-water management measures. The 'alternative' to wetland rehabilitation in this case, is effectively a combination of interventions, which, taken together, would reflect a higher 'replacement cost' and therefore rehabilitation benefit value. At the same time however, a combination of measures is likely to have a greater nitrogen reduction effect than the additional level of wetland retention gained with the rehabilitation. This raises the question of to what extent other kinds of capital can be considered replacements for ecological capital and, arguably, challenges a weak sustainability perspective.

This is also the case for the WWTW alternative. The relatively low volume wastewater treatment facility considered as the alternative to wetland rehabilitation has the capacity for a greater level of nitrogen abatement than the gain in potential achieved through wetland rehabilitation. While the value of the annual nitrogen retention benefit was estimated based only on the 'operating cost' of removal for an equivalent level of nitrogen retention as that gained with the rehabilitation, the benefit of avoided construction costs was a fixed amount associated with the capacity (size) of the WWTW rather than on the level of nitrogen removed. An option to resolve this inconsistency would be to spread the construction costs across the full nitrogen removal capacity of the WWTW (across its lifespan) to estimate a 'capital cost per unit of nitrogen removed' and then to combine this with the operating and maintenance costs per unit of nitrogen removed. While conceptually appealing, this approach doesn't reflect the 'reality' that in order to achieve an equivalent level of service as that associated with the rehabilitated wetland a WWTW of a capacity large enough to deal with the volume of flow would need to be built, thus incurring the full capital cost regardless of the actual amount of nitrogen removed.

Several additional nitrogen abatement options could also be considered. For example, autotrophic denitrification has been proposed as a potential alternative to standard heterotrophic denitrification (Zhou et al. 2011; Wang et al., 2018). In experimental research, autotrophic denitrification has been shown to be feasible for the removal of nitrate and nitrite, particularly from low concentration surface water, underground water, or wastewater treatment effluent (Zhou et al. 2011). However, at this stage in appears that heterotrophic denitrification remains more cost effective (Wang et al., 2018). In both cases, this form of treatment requires that the water to be treated be diverted through a treatment facility; which does not resolve the issue of a 'practical or realistic' alternative to wetland rehabilitation.

A potentially more realistic alternative to wetland rehabilitation is the use of floating wetland technology[89], which has the advantage of being applied without by-passing the stream flow (Pavlineri et al., 2017). Floating wetlands are, however, vulnerable to high velocity flows making practical implementation challenging and / or shortening their lifespan particularly in an event driven system. They are also vulnerable to damage by livestock and human interference; relevant risks in the case of the Mthinzima Stream context. Given the experimental nature of the technology, limited information is available regarding removal rates for instream application as well as maintenance costs and lifespan.

To compare alternatives to establish the level of service potential of each and to ascertain the 'least cost' alternative requires that a function for the relationship between the intervention and the outcome (effect on nitrogen pollution) be defined to estimate a marginal benefit for comparison against other alternatives. In this case, this would be particularly challenging for social initiatives aimed at reducing misuse of the sanitation system and improved monitoring of, and response to, sanitation system blockages. Least-cost determination requires a comprehensive analysis of alternatives; in the case of water quality enhancement, the 'alternative' to wetland rehabilitation likely to involve a combination of measures. For example, Grossman (2012) developed a cost minimisation model for nutrient abatement for the Elbe River (Germany) based on specific nutrient reduction targets to estimate a least-cost shadow price for wetland nutrient retention. The approach is data intensive, requiring data on the reduction of nutrients emitted by implementing a measure and the costs of implementing the measure for each alternative. In the case of Grossman (2012) the

[89] Floating wetlands are an experimental technology for improving the water quality of nutrient-rich waste and drainage waters. They consist of buoyant mats that are mass planted with emergent wetland plants and are anchored on the surface of the water body (e.g., anchored to bedrock or trees). Performance evaluation studies of floating wetlands indicate that they are effective in removing nutrients from urban storm-water runoff and ponded water bodies (Nichols et al., 2016).

information was extracted from an existing simulation model developed for the Elbe River system.

A reflection on the first two conditions of Shabman and Batie (1978) for reliable replacement cost valuation in the context of this case study leads to the question of what constitutes the 'same service' and, related, how realistic or practical must the alternative be in order to be considered an acceptable 'replacement'? In this case, while standard wastewater treatment removes nitrogen, WWTW infrastructure is suited to the treatment of point-source pollution and is impractical for the treatment of instream pollution. Wetland systems, on the other hand, may be particularly valuable in addressing non-point source pollution.

There are further 'divergences' between wetlands and WWTW that compound the complexity of the 'same service' condition. The rate of removal of nitrogen by wetland systems is influenced by a number of factors (see section 5.5.1) including environmental variables; it is possible that standard wastewater treatment may be less sensitive to such variables. However, conventional engineered wastewater treatment is also vulnerable to external factors and may 'fail' for example in the case of high flows. As noted by Heal et al. (2005:159) "both engineered and ecosystem approaches are vulnerable but they differ in the types of uncertainty associated with each investment". Both wetlands and WWTW experience a loss in nitrogen retention efficiency at threshold limits. In the case of WWTW, treatment efficiency declines as the volume capacity of the WWTW is reached and exceeded (Turpie et al., 2010). For wetland systems, nitrogen retention can be affected by prolonged high nutrient loading or loading rates in excess of critical thresholds which lead to a reduction in retention (Verhoeven et al. 2006). In the case of overloading, nitrates previously stored in the wetland system (e.g., through plant uptake) can be released into the system effectively becoming a nitrogen input to stream flow. Differences in threshold limits and conditions under which they are reached further influence the degree to which wetlands and WWTW can be considered alternatives.

Furthermore, wetlands and WWTW differ in the suite of water contaminants addressed. Both wetlands and WWTW are able to simultaneously address other water contaminants (such as phosphorus and pathogens). However, their capacity and comparability in this regard has not been considered in this case study due largely to a limited understanding of wetland processes and capacity across a range of other contaminants. In standard water treatment processes, the treatment of a range of contaminants is not an additive property (Graham, 2004; Rangeti, 2014; Turpie et al., 2017). In other words, water treatment costs are not an aggregation of the costs to treat individual contaminants, but are driven by a sub-set of contaminants of key concern. This means that the cost to address individual contaminants cannot be easily isolated, further complicating the valuation of wetland rehabilitation benefits through the replacement cost of nitrogen removal by WWTW (see Section 5.5.5 for further discussion). There are of course other co-services and benefits and potential disservices which differ between these two options which may not relate directly to the nitrogen abatement services, but may fundamentally affect the practicality, costs and benefits or each option and their overall comparability as alternatives. The replacement cost valuation approach doesn't make an allowance for such considerations.

Typically, the replacement cost method is considered suitable in valuing wetland ecosystem services related to water quality improvements where a more direct approach is not feasible (see section 5.3.3). For this case study, the search for a replacement alternative raised questions of whether there is a realistic alternative in this context. For this reason, the fourth consideration of reliable replacement cost valuation (Table 5.7) was assigned a 'moderate confidence' rating and it is likely that the replacement cost valuation understates the role of the wetland rehabilitation.

While commonly adopted as the alternative to ecosystem water quality enhancement, for this case, it is debateable whether the 'service' provided by a WWTW is similar to that provided through the wetland rehabilitation. In reality, the capacity of the WWTW alternative to remove nitrogen is likely greater than that achieved through rehabilitation of the wetland, thus overstating the value of the rehabilitation. On the other hand, from a practical perspective, an advantage of wetland water treatment is the ability to simultaneously address both point and non-point source inputs; only a combination of alternative measures can provide this same service. A WWTW is less able to address non-point source pollution (unless the stream were to be diverted through the WWTW), therefore understating the role of the wetland rehabilitation. Furthermore, the two 'alternatives' differ in their associated co-benefits. In this case, the divergent drivers of water pollution in the system mean that efforts to address excess nitrogen inputs require a combination of ecological and built infrastructure interventions and a concomitant investment in social and human capital. Factor 5 of the considerations for reliable replacement cost valuation is assigned a 'low confidence' rating in this case.

5.5.5 The value of the replacement - nitrogen abatement cost

An accurate estimate of the marginal replacement cost, in this case, the nitrogen abatement cost associated with a relatively small conventional WWTW, is fundamental to the reliability of replacement cost valuation. Reliability is improved when it is possible to derive the cost of the replacement from actual expenditure (Brouwer et al., 2013) or it can be confidently approximated for the local context (Factor 6, Table 5.7).

The marginal cost of nitrogen abatement was derived from an analysis of the operating and maintenance costs of standard WWTW undertaken by Turpie et al. (2010). The assumptions and limitations of the Turpie et al. (2010) analysis are thus transferred to this study. A particular factor in this regard, as noted in studies of water treatment (Graham et al., 2012; Rangeti, 2014), is the correlation between the quantities of many of the constituents / pollutants of water (e.g., nitrogen, phosphorous, suspended solids) and the fact that water treatment processes address (treat) several constituents simultaneously making it difficult to isolate the marginal costs of removal of a single constituent (Turpie et al., 2010). Furthermore, a treatment plant may be designed to target a particular constituent influencing both the capital and operating and maintenance cost of the plant (UNEP, 2015). In this way, as noted by Turpie et al. (2010), given that the treatment works studied were targeted primarily towards phosphorous removal, the nitrogen removal cost may be overestimated as it is possible that a facility designed purely for the removal of nitrogen could be less costly.

Confidence in transferring the estimated nitrogen removal cost from the Turpie et al. (2010) study is reduced in that the WWTW analysed were from Western Cape, many of which were operating above capacity and, therefore potentially less efficiently. This inefficiency could have influenced the estimated nitrogen removal rate used to derive the nitrogen removal cost. While operating and maintenance costs for WWTW in South Africa were available from other sources (e.g., Naidoo et al., 2016) these did not include a corresponding analysis of nitrogen removal rates of the treatment works to generate a cost per unit nitrogen removed. The cost of wastewater treatment can vary considerably across WWTW depending on the technology type and treatment capacity, among other factors (UNEP 2015; Naidoo et al., 2016). The costing of wastewater treatment is itself a complex task even before attempting to apportion cost to the removal or treatment of a single constituent. Turpie et al. (2010) addressed this latter issue by attributing the average cost of wastewater treatment to nitrogen removal and acknowledged that this assumption may have resulted in an overestimate of nitrogen removal cost.

Given these uncertainties, Factor 6 in the consideration of the reliability of the replacement cost value estimates was assigned a 'moderate confidence' rating for this case study. It is not conclusive that the replacement cost estimate accurately reflects local conditions (operation and maintenance costs of local WWTW) and there is uncertainty regarding the components that should be included in the replacement cost estimate (i.e., capital expenditure) with different approaches taken in the literature. Including capital expenditure in the replacement cost estimate, compared to a cost estimate based only on operation and maintenance costs, had a significant effect on the CBA results and the net welfare effect outcomes.

5.5.6 The demand for additional nitrogen removal

A specific weakness of the replacement cost method is that the approach does not directly address the demand for the benefit and associated service. The valuation is based on an assumed demand and, further, that the service or benefit would still be demanded if provided by the 'replacement' alternative. For the valuation to be reliable, there should be reasonable evidence to support the assumption (Shabman and Batie, 1978); this consideration is reflected as Factor 7 in Table 5.7.

The replacement cost method was applied in this case study under the assumption that any additional water quality enhancement within the Midmar Dam catchment would be fully demanded, in other words, that avoiding eutrophication of the Dam is a public preference (i.e., that the social value of clean water in Midmar Dam exceeds the costs of maintaining the water quality of the Dam). There are several arguments to support this assumption, as outlined in section 5.3.3. To reiterate briefly, these are primarily related to: national priorities to address the eutrophication of freshwater resources (DWS, 2018) and the adoption of a precautionary approach in the management of waste water (National Water and Sanitation Master Plan, DWS, 2018); and, at the local level, the social value of the clean water of Midmar Dam from both a water provisioning and recreational use perspective (Vundla et al., 2017) as reflected in the identification of ‘a reduction in water pollution into Midmar Dam, notably nutrient inputs associated with sewage contamination of the Mthinzima Stream’, as a priority management action for the district (WRC, 2002; GroundTruth 2012; Jogiat, 2013; Umgeni Water, 2019). The assumption that any gain in nitrogen reduction potential upstream of Midmar Dam is fully demanded is considered reasonable given these national and regional objectives and that the Mthinzima Stream is a major source of nutrients and pollution to Midmar Dam (Namugize et al., 2018). Factor 7 of the considerations for reliable replacement cost valuation has been assigned a rating of ‘high confidence’ for this case study.

However, a caveat should be noted. It has been assumed, based on an extensive existing knowledge base, that a reduction in nitrogen input to the Dam equivalent to that gained due to the wetland rehabilitation would translate into reduced risk of eutrophication. Limnological studies of the Dam to improve our understanding of the internal biological and chemical functioning of the impoundment and modelling the risk of eutrophication under different scenarios is an area of future research that would enrich the economic valuation of the wetland rehabilitation.

5.5.7 Partial analysis

The case study set out to assess the value of the wetland rehabilitation from the perspective of the added value associated with the primary objective of the rehabilitation to enhance the nitrogen removal potential of the wetland. The results provide a platform from which to gauge whether the rehabilitation is justified from an economic efficiency criterion based solely on achieving the anticipated nitrogen retention outcome. While responding directly to the priority rehabilitation outcome, the analysis results can only be viewed as a partial assessment of the value of the rehabilitation, limited to a single outcome (stream nitrogen abatement), TEV component (indirect use value) and value dimension (instrumental value). Consideration of any additional likely outcomes of the rehabilitation (positive and negative) and the distribution of these potential ‘benefits’ and ‘dis-benefits’ across differ groups enhances the usefulness of the ‘partial analysis’, by placing the economic value estimate(s) within the broader context of outcomes and values. This consideration is reflected as Factor 8 in Table 5.7.

From the TEV framing perspective, this case study analysed a single indirect use value associated with the rehabilitation. Additional indirect use values are likely to stem from the improved potential of the wetland for erosion control and carbon storage as determined by the ecosystem services assessment (GroundTruth, 2015). These ‘use’ values accrue to those directly benefitting from the increased service potential; in the case of erosion control, the land owners / users and in the case of carbon storage (climate regulation) broader society. Further to the present potential use value of the rehabilitation is the likelihood of increased future demand for water quality enhancement services within the Stream system – which can be classified as ‘option’ value as described earlier in the discussion. One aspect associated with future demand is the risk mitigation role of the rehabilitated wetland during the construction and initial operation of the proposed Mpophomeni WWTW.

There is also the likelihood of an appreciation for the increase in the integrity of the wetland and its potential to provide services by those not interacting with the wetland or its services (i.e., by those not benefitting from ‘using’ the wetland) in the form of non-use values related to the satisfaction that the wetland and its services are available to others, both currently and into the future. In a research study of citizen science using the Mpophomeni Settlement as a case study, responses from interviewees reflected non-use values in emphasising the importance of community well-being including that of future generations associated with aquatic ecosystem protection. The improved integrity of the wetland through rehabilitation would also contribute to biodiversity maintenance goals, for which there is a broader social appreciation beyond those interacting with the wetland or the ecosystem services it provides.

From the perspective of a broader conceptualization of values, additional value domains beyond the economic (instrumental) value framing were not considered or articulated through this partial valuation approach. While the dominant discourse on the value of wetlands and ecosystems in the upper uMngeni Catchment centres on the need to improve and protect water quality for human use (Sutherland et al., 2019), and therefore the instrumental value of wetland rehabilitation and maintenance, there is anecdotal evidence of values associated with *intrinsic, fundamental and eudaimonistic domains*[90]. Kolbe (2014:106) noted that "some respondents identify intrinsic environmental worth as being a key incentive for participating in citizen science". The responses of these participants reflect a sense of moral duty towards nature: "*They also deserve life just like us*" and "*there are animals there that need to be taken care of*". Research by Rivers-Moore (2016), which involved engagement with the members of the uMngeni Catchment Management Forum[91], revealed values associated with 'sense of place' evident in local residents' participation in wetland and river clean-ups and stewardship actions, and the value of wetland rehabilitation in building ecological resilience.

On the other hand, there could be undesired effects or 'dis-benefits' and trade-offs associated with the rehabilitation and preservation of the wetland which may influence the decision context and / or require additional measures to address. In this case for instance, grazing of livestock within the wetland area may be reduced through the proposed rehabilitation as a result of greater retention of water in the landscape affecting accessibility, particularly in the wet season (GroundTruth, 2015). This is an important aspect given that the Mthinzima wetland is located on communal land used primarily for grazing. However, wetlands as a source of livestock fodder tend to be of greater value in the dry season when other fodder is minimal (Pollard et al., 2008), at which point the wetland habitat is likely to be more accessible. There may also be a trade-off between use of the wetland for water quality enhancement and the conservation of biodiversity. While the rehabilitation of the wetland is anticipated to increase species diversity and promote biodiversity with the wetland system (GroundTruth, 2015), prolonged nutrient loading affects species composition and enriched wetland systems can lose species (Verhoeven et al., 2006).

[90] Following the heuristic framing suggested by Arias-Arévalo et al. (2018).

[91] The Upper uMngeni Catchment Management Forum is a voluntary forum active in the Upper Catchment and has a wide representation of organisations from provincial and local government, Umgeni Water (water supplier), NGOs and civil society organisations (including conservancies), the private sector, research institutions and individual residents.

The distribution of these potential ‘benefits’ and ‘dis-benefits’ across different groups is a crucial consideration. In this case study, the primary objective of the rehabilitation to enhance the quality of the water entering Midmar Dam from the Mthinzima stream produces benefits for the local and regional population in terms of potable water benefits and local, regional and international communities in terms of recreational benefits. The potential dis-benefit of reduced access to grazing during the wet season affects the immediate population, who are less likely to benefit from the downstream water-quality benefits. However, with continued degradation of the wetland, particularly further erosion / incision of the stream channel, ecosystem services associated with fodder provision will also decline (e.g., moisture retention into the dry season) reducing the grazing potential of the wetland area. In this way, local livestock owners stand to lose ‘access’ to wetland fodder not only in the wet season, but also during the dry season when this access is arguably more important.

This discussion emphasizes the ‘partial’ nature of the assessment in terms of both additional potential benefits (e.g., erosion control) and ‘dis-benefits’ (reduced access to grazing) of the rehabilitation and points to other relevant economic value components (e.g., option value) and value dimensions (e.g., intrinsic value). However, this consideration relied on secondary information and no specific or structured investigation was undertaken to ‘uncover’ additional benefits and values (e.g., through engaging with local residents). As such, Factor 8 of the considerations influencing the reliability or usefulness of the assessment has been assigned a ‘moderate’ rating.

Recognition of this analysis as ‘partial’ and the potential of a wider array of outcomes and sources of value of the wetland rehabilitation is not to imply that the ideal would be that all the value components of the TEV framework, or any other value framework, should be quantified and aggregated into a single total value to ‘complete’ the assessment. Rather, this acknowledgement aims to signal to decision-makers / evaluators that there may be other motivations in support of the wetland rehabilitation and dis-benefits of the rehabilitation, that should also be taken into account in considering the value of wetland rehabilitation and whether its ‘cost’ is justified. As such, the replacement cost method should not be used alone, or without careful consideration of the factors outlined in Table 5.7, to make a judgement on the value of the proposed rehabilitation or inform a decision on whether the rehabilitation should go ahead.

5.6 CONCLUSION

This case study has demonstrated an application of the replacement cost method to value the anticipated gain in nitrogen retention potential of a channelled valley bottom wetland as a result of a proposed rehabilitation intervention. The method is commonly applied in decision-making in environmental economics as it is cost effective and the results can be used in a CBA to generate a clear economic cost-benefit decision rule. In addition to presenting the application of the method to a case study, this chapter reflects on several of the study assumptions and challenges encountered from which a more generalisable set of factors influencing the reliability and decision-making power of the method has been derived.

The estimated economic value and corresponding CBA indicates that an economic efficiency gain is achieved when the projected costs of the construction and operation of a standard low volume WWTW are used as the replacement value for the rehabilitation. The estimated value is likely understated as a conservative wetland nitrogen reduction rate was applied in the analysis and the counterfactual scenario assumed that without rehabilitation the wetland would remain in its current state, whereas it is likely that the wetland would continue to degrade. On the other hand, the estimated value is likely overstated in that the capacity of the replacement alternative to remove nitrogen may be greater than that achieved through the rehabilitation. However, from a practical standpoint, a WWTW is less able to address non-point source pollution than a wetland, therefore understating the role of the wetland rehabilitation. The valuation is based on a single indirect use value (nitrogen reduction) which aligns with the primary objective of the rehabilitation, but does not capture or articulate the broader range of potential roles and values of wetland rehabilitation. This reality, which is likely of all valuation assessments, along with several sources of uncertainty, mean that the simple cost-benefit decision making rule should not be applied in this case without careful consideration of the study assumptions and limitations.

A strength of the assessment stems from the attention to assigning an economic value to the anticipated biophysical change and corresponding difference in nitrogen reduction potential as a result of the proposed rehabilitation. In this way, the case study adds to the currently small body of studies valuing a change, or difference, in ecosystem service flows (Chan and Satterfield, 2020). While the reliability of the results is reduced in this case by a paucity of local empirical studies of wetland nitrogen dynamics and additional uncertainties, a critical reflection on the case study revealed a number of aspects (Table 5.7) which, when carefully addressed, could improve the reliability of the replacement cost method in the evaluation of wetland rehabilitation. Even so, the replacement cost method should not be used on its own to evaluate the rehabilitation outcomes or as a basis for decision-making on whether the rehabilitation should go-ahead.

The findings emphasize that the economic valuation of wetland rehabilitation is highly context specific and relies on a detailed / comprehensive assessment of realistic differences in wetland (biophysical) function and ecosystem service flows between the 'with' and 'without' rehabilitation scenarios. So too, is this the case for the replacement cost method, which is often considered a conceptually simpler and cheaper approach relative to other economic valuation methods (Dickie, 2003). Application of the replacement cost method still requires an understanding of the study context from both a biophysical and social perspective, and measurement of the biophysical response and human relevant outcomes as a result of the rehabilitation which can be a complex undertaking and source of significant uncertainty depending on the extent of knowledge of the functioning of the wetland as a socio-ecological system. An interrogation of factors affecting wetland nitrogen removal underlined the likely limitations of a benefit transfer approach in quantifying the difference in ecosystem service potential as a result of wetland rehabilitation, particularly in the absence of local primary studies.

The application of the replacement cost method highlighted the complexity of identifying a suitable alternative to wetland rehabilitation, particularly one that meets the conditions of providing the 'same service' at the least cost (as specified by Shabman and Batie, 1978). Consideration of possible alternatives for use as the 'replacement' revealed that the divergent drivers of water pollution in the system require that efforts to address excess nitrogen inputs integrate a combination of ecological (e.g., wetland rehabilitation), built infrastructure (e.g., new wastewater treatment infrastructure) and governance (e.g., improved solid waste management) interventions and a concomitant investment in social and human capital (e.g., behavioural change). This finding suggests that wetland rehabilitation should be viewed as a complement, or co-investment, in a multipronged approach to addressing water pollution, and challenges, in this case, the weak sustainability position that different forms of capital are essentially substitutable in providing welfare.

Addressing the paucity of local empirical studies of wetland nitrogen dynamics and exploration of scenario analysis to moderate uncertainty are areas of future research that could strengthen the replacement cost approach in evaluating wetland rehabilitation for nitrogen abatement. Sensitivity analysis, which is typically employed in CBA to explore the effect of key assumptions on the results, may not be sufficient for gauging the influence of alternative futures on the value of wetland rehabilitation. Scenario analysis moves beyond this limitation as a method for developing and evaluating possible futures to account for uncertainty and complexity (Khosravi and Jha-Thakur, 2019). Consideration of a range of scenarios (e.g., possible futures) would add value to the economic analysis as a decision-aid.

Having considered the uses and limitations of quantitative, market-based valuation techniques, the following chapter presents key insights from three additional evaluation case studies where monetary-based valuation was considered, but found to be not possible or appropriate, and alternative approaches were applied to explore other options for articulating the contribution and value of wetland rehabilitation.

CHAPTER 6: INSIGHTS FROM NON-MONETARY EVALUATION CASE STUDIES

The point of departure for the research case studies was biophysically grounded marginal economic valuation employing a monetary metric. However, it soon became apparent that for many of the projects selected for outcomes evaluation, such valuation was not feasible or appropriate (i.e., given the aims of the rehabilitation). Key findings of three non-monetary evaluation case studies are presented here as they demonstrated some of the challenges encountered in attempting to apply economic valuation methods to wetland rehabilitation projects in South Africa and explored other approaches to evaluating social benefit outcomes. The case study experiences provide insights for consideration in suggesting an approach or framework for the valuation of wetland rehabilitation outcomes which is further developed in Chapter 7.

6.1 EDENDALE MALL WETLAND: ECOLOGICAL VALUES

An evaluation of a wetland rehabilitation associated with the development of an urban shopping mall (the Greater Edendale Mall, Pietermaritzburg, KwaZulu-Natal)[92] highlighted the importance of a common understanding of information requirements between ecologists and valuers. The case study is also valuable in that it draws attention to 'demand' and 'access' as critical dimensions in evaluating the importance to people of wetland rehabilitation. A potential pitfall of valuation studies, particularly those adopting an ecosystem services framing and a value transfer approach, is to interpret a change in the biophysical capacity of a wetland to supply a service directly as a valued benefit.

With regard to the first point, baseline and post-rehabilitation water quality monitoring records were expected to provide an opportunity for biophysically grounded marginal valuation based on measured, rather than estimated, differences in water quality

[92] This case study was part of the Water Research Commission Project K5/2344. The valuation assessment was undertaken by M Browne and is detailed in project deliverables 8 (2016) "A detailed evaluation of the infield observations, monitoring and assessment of the outcomes of the rehabilitation at the Greater Edendale wetland" and the ecological evaluation is detailed in project deliverable 9 (2018) "An integrated report detailing the monitoring and evaluation of the rehabilitation undertaken at the Greater Edendale Mall and Manalana wetland sites". The case study is also discussed in the final project report (Walters et al., 2019) available from the South African Water Research Commission.

enhancement. However, the data only extended to records of the concentration of specific contaminants and did not include flow measurements. This precluded the calculation of actual pollutant load reductions (e.g., kilograms of nitrogen removed) by the wetland to inform the economic valuation. Concentration data alone are not sufficient for quantifying the amount of pollutant removed from the water by the wetland. While the economic valuation could have been conducted based on modelled flows or pollution retention rates drawn from the literature, given the highly modified setting of the wetland the results would be of relatively low confidence.

As noted by Heal et al. (2005), the definitions and measures of ecosystem services need to be similarly defined across both ecological and social perspectives. This is equally important in the design and implementation of monitoring to generate information that supports the evaluation of rehabilitation outcomes from multiple perspectives. Given the expense of comprehensive monitoring, it is not reasonable to expect every wetland rehabilitation project to implement extensive monitoring; in practice the extent of monitoring is typically guided by the rehabilitation objectives. This was evident in this case, where the objective of the rehabilitation was to ensure no-nett-loss of wetland integrity as a result of the construction of a shopping mall and monitoring focused on indicators of ecological integrity (GroundTruth, 2010). The objectives did not include the maintenance or enhancement of any specific ecosystem service or benefit, nor specify a particular goal related to human well-being. The availability of the water quality data was not directly related to the rehabilitation; it was a condition of the environmental authorization for the development to ensure downstream water quality was not negatively impacted. For this purpose, monitoring of pollutant concentrations against target water quality ranges is currently considered sufficient and flow monitoring is not specified.

Given the rehabilitation objectives and available data, the evaluation focused on the ecological outcomes of the rehabilitation and made recommendations to the landowner regarding maintenance activities to improve and sustain the ecological integrity of the wetland. Arguably, this was the primary management-relevant information as it linked directly to the aim of the rehabilitation. However, while not a specific objective, the rehabilitated wetland was intended to be an education and recreation feature within the mall property. This was reflected in the ecosystem service assessment results comparing the pre-rehabilitation and post-rehabilitation scenarios which indicated a significant increase in 'tourism and recreation' and 'education and research' services (GroundTruth, 2010).

During the evaluation, these services were investigated through semi-structured interviews with mall patrons. Overall, the responses indicated that patrons had little awareness of the functions and benefits of wetlands, some suggested alternative land-uses were preferred, while others had not realized that a wetland was present. The responses suggest that, while the potential of the wetland to provide 'tourism and recreation' and 'education and research' services increased with the rehabilitation, these benefits were not being realized or recognized. In this case, additional inputs related to awareness raising (e.g., signboards, school group and student tours) are needed for people to 'access' the education related service potentials. In terms of 'recreation' services, the findings tentatively suggest that people do not attribute recreational benefits to the wetland, as such the increase in potential to supply this service has no direct value in the current setting. This case study illustrates 'demand' and 'access' as critical dimensions in evaluating the meaning or importance to people of wetland rehabilitation.

6.2 XHARAS WETLAND: A RAPID ASSESSMENT APPROACH

The evaluation of a communal wetland in a semi-rural setting[93] illustrates an application of a rapid ecosystem service assessment, incorporating both supply and demand aspects, as an alternative, or pre-cursor, to economic valuation of rehabilitation outcomes. In this case, there was a need to consider the contribution of the rehabilitation across a range of ecosystem services, but resource limitations meant that it was not possible to quantify and value all the contributions in detail. This case study was important as it explored a relatively rapid and low resource intensive approach to assessing the human-relevant outcomes of wetland rehabilitation.

[93] This case study was part of the Water Research Commission Project K5/2344 and is detailed in project deliverable 13 (2018) "An integrated report detailing the evaluation of nine wetland rehabilitation sites within South Africa". The assessment was led by D Kotze following the valuation assessment design developed by M Browne, and with contributions by M Browne. The case study is also discussed in the final project report (Walters et al., 2019) available from the South African Water Research Commission.

Economic valuation of wetland rehabilitation can be a resource intensive and complex undertaking, limiting both the quantity of valuations undertaken as well as the number of ecosystem services and associated benefits assessed[94] (Laurans et al., 2013). Examples of wetland rehabilitation valuations in South Africa (e.g., Black et al., 2016 and the case studies described in Chapters 4 and 5) focus on a few key attributes. While this is a practical approach and particularly appropriate if the rehabilitation objective is linked to a single, or few specific, outcomes; when the rehabilitation is intended to improve or secure a variety of ecosystem services and contribute to human well-being more broadly, then a range of services and benefits need to be considered.

The rapid assessment applied in this case study was based on an expert judgment approach to score both the effect of the rehabilitation on the potential supply of ecosystems services and the demand for the affected services on a low to high scale for each service. Each 'score' was accompanied by a supporting rationale and indication of the level of confidence in the judgement. The scoring was informed by an assessment of the biophysical and social context of the site and the development of a conceptual model of how key attributes of the wetland affected the services supplied by the wetland and how these attributes (and, in turn, the supply of ecosystem services) had been changed by the rehabilitation interventions. This was guided by the indicators and methods outlined in the WET-EcoServices tool (Kotze et al., 2008) and drew on the results of wetland heath assessments undertaken prior to and post rehabilitation and water table monitoring. The demand for ecosystem services was assessed primarily through considering the proximity of the wetland to human settlement and accessibility of the wetland to humans; the land-use and land tenure upstream, within and downstream of the wetland; and the level of vulnerability and dependency[95] of the affected individuals.

---

[94] The review of wetland restoration valuation studies (Chapter 3), indicated that economic valuations were most often based on a single attribute of the wetland system, which could be a result of the objective of the intervention to improve or protect a particular service, and/or because of the costly and complex nature of such valuations.

[95] An indication of the level of vulnerability and dependency of the affected individuals was determined by considering the uniqueness of the wetland with respect to the supply of services; availability of, and access to, of other sites to supply the services provided by the wetland in question (for provisioning services this would generally depend on the availability of another similar wetland/s nearby and accessibility in terms of tenure); availability of alternatives provided by built infrastructure / municipal services; and the socio-economic status of the wetland beneficiaries/users.

The results of the assessment suggested that the greatest contribution of the rehabilitation was to the maintenance of biodiversity. Demand for biodiversity maintenance was particularly high given the wetland's location in an area where wetlands have been identified as special habitats in need of conservation attention. The least significant contribution of the rehabilitation was to phosphate assimilation. While the rehabilitation interventions contributed significantly to the potential of the wetland to assimilate phosphorous, both the opportunity to mobilize the potential as a service and the demand for the service were low given that upstream inputs of phosphate were limited. Furthermore, there was no indication of any major increase in phosphate inputs in the upstream catchment, at least in the medium term.

This case study was important as it explored a relatively rapid and low resource intensive approach to assessing the human-relevant outcomes of wetland rehabilitation. This is useful when there is a need to understand the likely contribution of the rehabilitation across a range of ecosystem services, but resource limitations preclude comprehensive quantification and valuation. In addition, a rapid approach could be applied to a number of potential rehabilitation sites and the results used to rank the sites in terms of their relative contributions. To this end, the basic approach adopted here could be structured into a more formal index-based analysis using a numerical scoring approach similar to that proposed by Burkhard et al. (2012). Furthermore, a rapid ecosystem service approach could be used as a preliminary assessment to identify likely rehabilitation outcomes, benefits and values to inform the selection and design of more comprehensive valuation assessments.

The case study revealed several important limitations of the South African ecosystem service assessment tool (WET-EcoServices). Firstly, the tool inadequately accounts for describing the demand for ecosystem services. This illustrates the point made by Chan and Satterfield (2020) that the biophysical outcomes of ecosystem changes are often interpreted as changes in ecosystem services and benefits to people without attention to the social system and elements of 'access' and 'demand'. The rapid approach applied in this case study, took a step towards addressing this limitation by considering demand factors more explicitly. However, more attention to the dynamic nature of social systems and how the demand for potential ecosystem services and benefits is likely to change in the future is warranted. Applying the approach along with a scenario analysis process that considers a number of realistic 'possible futures' is one way to address this limitation. Further work could explore how dimensions of access, as well as the additional aspects of 'exposure' and 'co-production' suggested by Chan and Satterfield (2020) could be incorporated into rapid assessment approaches.

Secondly, in some instances, several attributes or resource types are grouped together under a single ecosystem service type in the WET-EcoServices classification. These may need to be separated and assessed individually. Thirdly, some ecosystem services are not included at all in the WET-EcoServices tool, but may be relevant in certain landscapes. For example, as revealed in this case study, groundwater recharge, previously excluded from the WET-EcoServices classification, may be a potentially important contribution of the many largely temporary sandy wetlands of the area. The classification warrants review in light of the findings of a growing number of applications. The latter two limitations point to a potential risk in adopting pre-defined ecosystem service / benefit classification systems which may 'miss' the nuances of individual sites and their particular contexts.

6.3 BAYNESPRUIT CATCHMENT: MULTI-CRITERIA ASSESSMENT

The case study of a proposed urban stream rehabilitation project, the Baynespruit catchment (Pietermaritzburg, KwaZulu-Natal) to enhance water quality[96] emphasizes the importance of an understanding of the site and management context in framing the evaluation and selecting methods appropriate to generating management-relevant information[97]. While economic valuation and a CBA of wetland rehabilitation had been suggested by stakeholders, consideration of the catchment and decision-making context revealed that it was not yet clear if and how wetland rehabilitation could be implemented and a different approach was warranted, at least initially.

[96] This case study was part of the Water Research Commission Project K5/2345 and is detailed in project deliverable 12 and the final technical report (Jewitt et al., 2020), which is available from the South African Water Research Commission. The evaluation was undertaken by M Browne with contributions from L Mugwedi and G Jewitt.

[97] Cook et al. (2013:669) define management-relevant information as being "salient (relevant and timely), credible (authoritative, believable, and trusted) and legitimate (developed via a process that considers the values and perspectives of all relevant actors) in the eyes of both researchers and decision makers".

Conceptually, rehabilitation of wetlands within the catchment could contribute to water quality enhancement; however, at the time of initiating the project, there was little information on the wetlands present within the catchment, their respective condition and functionality and whether any, or which, have a relatively high potential for rehabilitation to contribute to water quality enhancement[98]. Furthermore, there are a number of drivers of poor water quality and ecosystem degradation within the catchment and continuation of these drivers are a threat to the long-term sustainability of rehabilitation efforts. In the context of a small urban catchment, wetland (and ecosystem) rehabilitation is itself a form of land-use adding to the 'competition for space' and potentially creating conflicts between residents. It became evident that there is an interdependent relationship between ecosystems, society, and water security within the catchment and stream rehabilitation efforts need to extend beyond ecological interventions (e.g., wetland rehabilitation) to include investment in built infrastructure and social and human capital[99].

In this case study, consideration of the catchment and management context revealed that exploration of a range of intervention options and the interactions between their outcomes was warranted, along with additional ecological assessments, before committing to a detailed CBA of wetland rehabilitation. To this end, a broad evaluation was undertaken involving expert consultation and a multi-criteria assessment as a first step towards identifying and prioritizing intervention options[100]. A sub-set of options was then assessed in terms of how each could be implemented in the catchment, the associated implementation costs (financial estimate), the likely water security outcomes, any co-benefits and dis-benefits and to who these would apply, and any potential risks or constraints. Given the range of intervention types (ecological, technical, social) and the connections between the biophysical, social, political, economic and governance dimensions within the catchment, multiple perspectives

[98] There were no wetland rehabilitation plans identifying specific sites and detailing interventions and limited information for assessing or predicting wetland response and biophysical change as a result of rehabilitation, and calculating rehabilitation costs.

[99] A stakeholder workshop involving the municipality and local experts posing the question of 'what to do in the catchment to improve water quality' formed the basis for identifying the challenges and intervention options within the catchment.

[100] Intervention options identified through expert consultation and a review were assessed drawing on the framework of Depietri and McPhearson (2017); the criteria considered were reliability, long-term durability or resilience, reversibility and flexibility, feasibility, relative cost, co-benefits and dis-benefits.

and mixed methods (i.e., a range of detailed biophysical, social economic and development studies) and integration of the resulting, disparate, information are needed to evaluate water security interventions within the catchment (Jewitt et al., 2020).

An understanding of the context is integral to identifying appropriate methods and framing detailed studies towards generating management-relevant information. This was evident in this case study which highlighted that additional studies were needed before economic evaluation and CBA of wetland rehabilitation could generate salient and credible information; and that such information would need to be integrated with the findings from other approaches in the evaluation of water security interventions.

In addition, contextualisation of the system highlighted aspects important for any future evaluation of wetland rehabilitation outcomes. The 'space' limitations of the urban setting, and the varied, sometimes conflicting, land-use objectives suggest that opportunity cost would be an important consideration. The existing drivers of wetland degradation in the catchment suggest that rehabilitation interventions in this case may have higher monitoring and maintenance requirements (and therefore long-term costs) to ensure wetland sustainability. The socio-economic and land-use context points to significant differences across groups of people in terms of demand for, and access to, wetland ecosystem services and who is likely to benefit from wetland rehabilitation and who may lose. Given the heterogeneity in socio-economic, cultural and livelihood vulnerability factors across residents of the catchment, comparing the 'significance' of benefits and losses to different people using a monetary metric alone will be inappropriate.

These three case studies demonstrated challenges encountered in attempting to apply economic valuation methods to wetland rehabilitation projects in South Africa and explored other ways of articulating the contribution and importance of the rehabilitation. The Edendale Mall wetland assessment illuminated data challenges and drew attention to the need for collaboration between wetland specialists and valuers to ensure a common understanding of information requirements. The challenges encountered suggest that rehabilition monitoring needs to be designed with benefit valuation in mind, if such valuation is desired. The Baynespruit Stream case illustrated constraints and risks to the ability, and long-term sustainability, of wetland rehabilitation to address the water security objectives of the catchment, highlighting the importance of attention to site specific factors in considering whether to invest in wetland rehabilitation. The case study demonstrated a multi-criteria evaluation approach to comparing various intervention options, which could be used at a high level to 'screen' a number of options for achieving specific obejectives before undertaking detailed evaluations. The Xharas study highlighted the practical reality in rehabilitation practice in South Africa of limited time and resources to conduct comprehensive valuations. To this end, the case study explored a relatively rapid and low resource intensive approach to assessing the human-relevant outcomes of wetland rehabilitation, which could be further developed in future applications.

The following chapter integrates the case study experiences with the findings of the theoretical research components to propose a guiding framework for the valuation of wetland rehabilitation to achieve the second, and final, aim of the research. The chapter closes with a reflection on 'where to next' for wetland rehabilitation valuation in conclusion of the thesis.

CHAPTER 7: RESEARCH SYNTHESIS AND CONCLUSIONS

This research set-out to explore the economic valuation of wetland rehabilitation with the intention of quantifying and valuing the outcomes of a sample of wetland rehabilitation projects in South Africa and providing an approach for generating estimates of the value of wetland rehabilitation for use in South Africa. These aims were in response to the dearth of studies evaluating the economic value of wetland rehabilitation (Chapter 3) and increasing calls for outcomes evaluation to support the growing practice of wetland rehabilitation in South Africa (Chapter 1). Intended to address a research need in South Africa, the resulting framework is, however, more generally applicable, especially in, but not limited to, developing country contexts.

While initially grounded in the field of mainstream economics, the research led into a number of fields including philosophy and axiology, social-ecological systems and social-ecological relations (political ecology) perspectives, a number of environmental science areas and ecosystem services research, and livelihood and human well-being frameworks. A deeper look into economic theory itself revealed an evolution of thinking on the meaning of 'value' and the economic view of 'nature'; thinking which continues to evolve as evidenced in the extensive literature arguing that the neoclassical economic model is limited as an explanation of human behaviour and too restrictive in its conceptualisation of value and the divergence of multiple schools of thought within the economics discipline (Chapter 2). While perplexing to try and disentangle, the numerous concepts of, and debates about, environmental values, ecosystem services, and economic valuation serve to emphasize the diverse ways people think about, connect with and value wetland systems, and, therefore their rehabilitation. Ultimately, the value of wetland rehabilitation cannot be reduced to a single 'value' measure elicited with a single method. While the economic value perspective can be a useful framework for considering the value of wetland rehabilitation, it is a partial one, as is the case for any single perspective.

This chapter synthesizes the research findings in respect of the research aims, presents a proposed framework for the valuation of wetland rehabilitation in South Africa derived from theory and the practical case studies, and closes with a reflection on 'where to next' for wetland rehabilitation valuation in conclusion of the thesis.

7.1 RESEARCH SYNTHESIS

7.1.1 Case studies of the value of wetland rehabilitation in South Africa (aim 1)

The aim of quantifying and valuing the outcomes of a sample of wetland rehabilitation projects in South Africa was achieved through the evaluation of five wetland rehabilitation projects. A synopsis of the case studies is presented in Table 7.1. The results of the case studies, except in the Baynespruit Stream case, provide supporting evidence and motivation for the investment in wetland rehabilitation in South Africa, demonstrating the achievement of outcomes related to the rehabilitation objectives and identifying, and measuring in some cases, values associated with the rehabilitation objectives.

The Baynespruit case illustrated constraints and risks (lack of space, severe water pollution inputs and conflicting social objectives) to the ability, and long-term sustainability, of wetland rehabilitation to address the water security objectives of the catchment, highlighting the importance of attention to site specific factors in considering whether to invest in wetland rehabilitation. The case studies also illuminated several challenges and aspects, which provide insights for proposing an approach to the valuation of wetland rehabilitation in South Africa.

The Mthinzima case study (Chapter 5) demonstrated the indirect use value of the anticipated gain in the nitrogen retention service of the wetland as a result of the proposed rehabilitation. Based only on this value, which aligns with the primary objective of the rehabilitation, the corresponding cost-benefit analysis indicated that an economic efficiency gain would be achieved. However, several sources of uncertainty suggested that the simple cost-benefit decision making rule should not be applied without careful consideration of the study assumptions and limitations. In this respect, a critical reflection on the application of the replacement cost method in the case study revealed a number of aspects which, when carefully addressed, could improve the reliability of the replacement cost method in the evaluation of wetland rehabilitation. The case study emphasized that, even in the case of what is considered a relatively straight forward valuation method, an understanding of the theory underlying the method and careful interpretation of the results are necessary.

The Manalana wetland case study (Chapter 4) demonstrated the direct (consumptive) use value of the contribution of wetland rehabilitation to three wetland-based livelihood activities. The corresponding cost-benefit analysis generated benefit-costs ratios in close proximity to a value of one, both above and below one depending on the discount rate, indicating that a conclusion regarding an economic efficiency gain is sensitive to small changes in the benefits, values, and costs included and the chosen discount rate. While the valuation aligned with the primary objective of the rehabilitation – to secure the livelihood benefits of the wetland – several values associated with these benefits were revealed during the valuation process, specifically food security, social safety-net and option values, which were not reflected in the economic valuation. The case study emphasized the partial nature of an assessment applying a single valuation method or value perspective and highlighted the need for interpretation of the results with regard to the 'missing' values and the particular context of the beneficiaries. In this case, the partial nature of the valuation could lead to a poor conclusion on the value of the rehabilitation should the cost-benefit results be considered without careful interpretation and recognition of the 'missing' values and socio-economic vulnerability of the beneficiaries.

These cost-benefit assessments draw attention to the influence of discounting and the choice of discount rate on the results, in particular, the Manalana case benefit-cost results were sensitive to the choice of discount rate. In the context of ecosystem valuation, discounting and the appropriate discount rate is a subject of debate with arguments for discounting the future benefits that flow from ecosystems at a lower rate, or not at all (e.g., Fenichel et al., 2017). Rather than applying a lower discount rate, Mullins et al. (2014) proposed that the usual discount rate for public investments be applied to environmental projects, but that a hyperbolic discount function be applied instead of an exponential function. These aspects warrant investigation in future applications and in the context of what is appropriate for South Africa.

Table 7.1: Synopsis of case studies

	Manalana	Mthinzima	Baynespruit	Xharas	Edendale
Rehabilitation goal	Maintain livelihood support activities	Enhance water quality	Enhance water security related services	Generally improve ecosystem services	Compliance with legislation No-nett-loss of wetland habitat
Timing of evaluation	Post-rehabilitation	Pre-rehabilitation	Pre-rehabilitation	Post-rehabilitation	Post-rehabilitation
Location context and beneficiary group	Peri-rural communal area Wetland farmers and members of the local village	Downstream of a peri-rural settlement, upstream of a priority water supply dam Downstream water users	Urban stream Local residents (across different socio-economic groups), downstream water users	Communal area characterised by low economic activity and high unemployment Broader public, local livestock owners	Urban Private retail developers Public benefit - critical habitat maintenance, water quality enhancement
Outcome focus	Rural livelihood support Provisioning ecosystem services	Water quality enhancement benefit Regulating ecosystem services	Water quality and stream flow regulation benefits	Maintenance of biodiversity and landscape connectivity, water and fodder for livestock (provisioning services)	'No-nett-loss' of wetland habitat
Valuation approach / method	Economic valuation Market-based method	Economic valuation Replacement cost method	Multi-criteria evaluation	Rapid ecosystem service supply and demand (expert judgement supported by context & system analysis)	Biophysical indicators, stakeholder perspectives (interviews)
What was 'measured'	Direct use value Monetary metric	Indirect use value Monetary metric	Potential contribution of the rehabilitation across a range of criteria	Indication of use values and ecological values	Biophysical outcomes – equivalent area of wetland habitat
Metric	Monetary	Monetary	Scores, narrative	Scores, narrative	Biophysical metrics, narrative
Scale	Local – adjacent village	Catchment – downstream water users	Local – adjacent residents, downstream water users	Local households, global public	Catchment

	Manalana	Mthinzima	Baynespruit	Xharas	Edendale
Interpretation	Partial valuation - missing safety-net value, option value Cost-benefit analysis – justify investment, demonstrate objectives achieved	Partial valuation Cost-benefit analysis - motivate investment (funding)	Compared wetland rehabilitation to a range of water security related intervention options, proposed a continuum of ecological to built infrastructure interventions	Demonstrate objectives achieved - rapid approach, mix of high and low confidence 'scores'	Demonstrate objectives achieved Recommendations for management of the wetland from an ecological perspective
Key learnings	Emphasised the importance of interpreting the value estimates in the context within which they were derived	A set of factors that should be considered in the application of the replacement cost method to improve the reliability of the results	Highlighted the importance of the conceptualisation process in identifying conflicting preferences across actors and information gaps (lack of clearly defined alternatives) and need for consideration of other intervention options	Explored a relatively rapid and low resource intensive approach in the context of limited resources for comprehensive assessment The approach could be further developed into a systematic rapid assessment tool	Highlighted the importance of alignment between biophysical data and valuation needs

The additional three case studies highlighted challenges in the application of economic valuation and demonstrated other ways of articulating the contribution and importance of the rehabilitation. The Edendale Mall wetland assessment illuminated data challenges in defining and quantifying the difference in water quality enhancement service as a result of the rehabilitation. While a lack of biophysical data is recognized as a common challenge to economic valuation, in this case – a post-rehabilitation assessment supported by long-term water quality monitoring – rather than a lack of data, the challenge was a 'mis-alignment' between the 'ecology' and the 'economics' with respect to the type and form of data required. Data challenges were similarly experienced in the Mthinzima case, where the lack of primary studies of nitrate retention by wetlands in South Africa meant that retention rates derived for wetlands in other countries were used, thus reducing confidence in the results. These case studies highlighted that monitoring needs to be designed with benefit valuation in mind, if such valuation is desired, and drew attention to the need for collaboration between wetland specialists and valuers to ensure a common understanding of information requirements.

This is not to suggest that every wetland rehabilitation project should include comprehensive, benefit valuation appropriate monitoring; in practice the scope of monitoring and evaluation is guided by the rehabilitation objectives and with the aim of providing management relevant information. This was demonstrated in the Edendale Mall case, where the rehabilitation aim did not specify a particular ecosystem service or benefit and biophysical assessments were sufficient to provide the required management information. However, a sample of representative comprehensive monitoring and post-rehabilitation valuation studies are needed to provide a base for pre-rehabilitation predictive studies and from which benefit and value transfer approaches can be explored as well as the development of more rapid valuation approaches or indicators.

The Xharas study highlighted the practical reality in rehabilitation practice in South Africa of limited time and resources to conduct comprehensive valuations, yet a desire of decision-makers and rehabilitation planners for some indication of the contribution and importance of the rehabilitation to people. To this end, the case study explored a relatively rapid and low resource intensive approach to assessing the human-relevant outcomes of wetland rehabilitation. The results of the assessment, based on a joint consideration of supply and demand, suggested that the greatest contribution of the rehabilitation was to the maintenance of biodiversity, given the wetland’s location in an area where wetlands have been identified as special habitats in need of conservation attention. Such rapid approaches could be particularly useful when there is a need to understand the likely benefits as a result of the rehabilitation, without need for rigorous value measurement, and could be used to compare and rank potential rehabilitation sites based are specific objectives. The basic approach applied here could be further developed into a more formal index-based analysis using a numerical scoring approach similar to that proposed by Burkhard et al. (2012).

The Baynespruit study emphasized the interconnection between wetland systems and the surrounding catchment in terms of the biophysical, social and institutional context. In this case, while wetlands are recognized as being able to provide water quality services, the nature of the pollution, limited physical space in the urban context to implement rehabilitation and the conflicting land-use needs in the catchment suggest that wetland rehabilitation may not be as effective in the specific context and pointed to the need to consider additional or alternative interventions to achieve the water management related objectives. To this end, the case study demonstrated a multi-criteria evaluation approach to identifying key objectives and comparing the potential contribution of various intervention options to the objectives. Such an approach could be used at a high level to first compare a number of options for achieving specific goals and prioritize options for further detailed investigation.

Several common insights emerged across the case studies. The case studies drew attention to ‘demand’ and ‘access’ and highlighted a potential pitfall of valuation studies, particularly those adopting an ecosystem services framing and a value transfer approach, is to interpret a change in the biophysical capacity of a wetland to supply a service directly as a benefit without considering whether there is, or will likely be, a demand for the additional supply or whether people are able to derive benefit from the biophysical changes. In the Edendale Mall case, such an approach would have led to overestimating the value of the rehabilitation with respect to recreation benefits, for which there appeared little demand by mall patrons, or consumptive benefits, to which there was no access. In the Mthinzima case, the construction of a wastewater treatment plant upstream of the wetland would likely reduce the demand for the additional water quality enhancement service of the wetland; while in the Manalana case declining local livestock production reduces the livestock fodder benefit of the wetland. Chan and Satterfield (2020) concluded that the biophysical outcomes of ecosystem changes are often interpreted as changes in ecosystem services and benefits to people without attention to the social system and elements of 'access' and 'demand'. On the other hand, the case studies also drew attention to future demand, option value and potential use (e.g., risk mitigation in the case of water quality enhancement services); elements which require further investigation including attention to how to deal with uncertainty around future potential use and if or how such values should be weighted relative to current or known use (similar to the rationale of discounting future benefits).

The case studies further revealed that benefit and value types emerged during the process of the valuation assessment, and / or that pre-conceptions about values and benefits were confirmed or challenged, specifically as an understanding of the catchment context was developed and through engagement with affected people and various stakeholders. Interviews with patrons of the Edendale Mall revealed limited aesthetic and recreation benefits associated with the rehabilitation. It was the representative of the local water services provider who suggested the importance of the Mthinzima wetland rehabilitation for risk mitigation and the residents of Craigieburn Village who highlighted food security, relation, option and bequest values associated with the rehabilitation of the Manalana wetland. This emphasizes the diversity of values that may be associated with a rehabilitation and, further, that a single benefit can be associated with several different types or dimensions of value. Within the TEV framing, the crop cultivation benefit of the Manalana wetland rehabilitation has direct consumptive use value, but also option value and bequest value; beyond the TEV framing, crop cultivation was also associated with relational values related to sense of identity and social standing of wetland farmers, who, being woman in a semi-rural and traditionally patriarchal society, may not otherwise have the same social standing.

The monetary value estimates presented in the case studies were subject to uncertainty, stemming from challenges in quantifying changes in the capacity of the wetland to supply a service or benefit (e.g., a lack of locally relevant nitrogen retention in rates the Mthinzima case, resource production yields under degraded wetland conditions in the Manalana case) and the valuation approaches. In the Edendale Mall case, data challenges constrained the application of economic valuation; in the Mthinzima case, constraints to modelling dam eutrophication relationships meant that an indirect valuation method was used (replacement cost) and further limited the scope of the valuation (from benefits of improved water quality for both consumption and recreational use to only consumption related benefits). Sources of uncertainty in the valuation methods related to the fulfilment of the conditions of the replacement cost method (Mthinzima case) and reservations in respect of whether the 'village prices' applied in the Manalana case to value crop production are an accurate reflection and the transfer of the annual direct use value of livestock from a possibly outdated study. However, as argued by Freeman (2003:19) our "understanding of the physical, biological, and socioeconomic systems that must be modeled to produce the relevant scenarios for welfare comparisons" is imperfect and some level of uncertainty is inherent. This again suggests the need for careful interpretation of the results, reflecting on the potential sources of uncertainty.

7.1.2 Guiding framework for the valuation of wetland rehabilitation (aim 2)

These case study experiences and insights from the theory are synthesized into a guiding framework for the valuation of wetland rehabilitation, Figure 7.1, to achieve the second aim of the research, which was to provide an approach for generating estimates of the value of wetland rehabilitation for use in South Africa. The resulting framework is, however, more generally applicable, especially in, but not limited to, developing country contexts. There are numerous existing frameworks for ecosystem valuation; the guiding framework presented in this thesis is not proposed as an alternative to these, but rather intended as a complement, with the aim of contextualizing the more general valuation concepts and 'steps' within the specific case of wetland change as a result of rehabilitation, and drawing attention to particular aspects of importance identified through the research. Inspired by the suggestion of Norton and Noonan (2007:672), the proposed framework offers a way to "start thinking in addition to economic analysis" (Norton and Noonan, 2007:672). The various elements of the framework are discussed in the following sections[101].

[101] A table defining each element of the framework is presented in Appendix 7.1. The table, along with several additional supporting materials included as appendices, could be further developed into a manual or step-by-step guide in support of the proposed valuation framework.

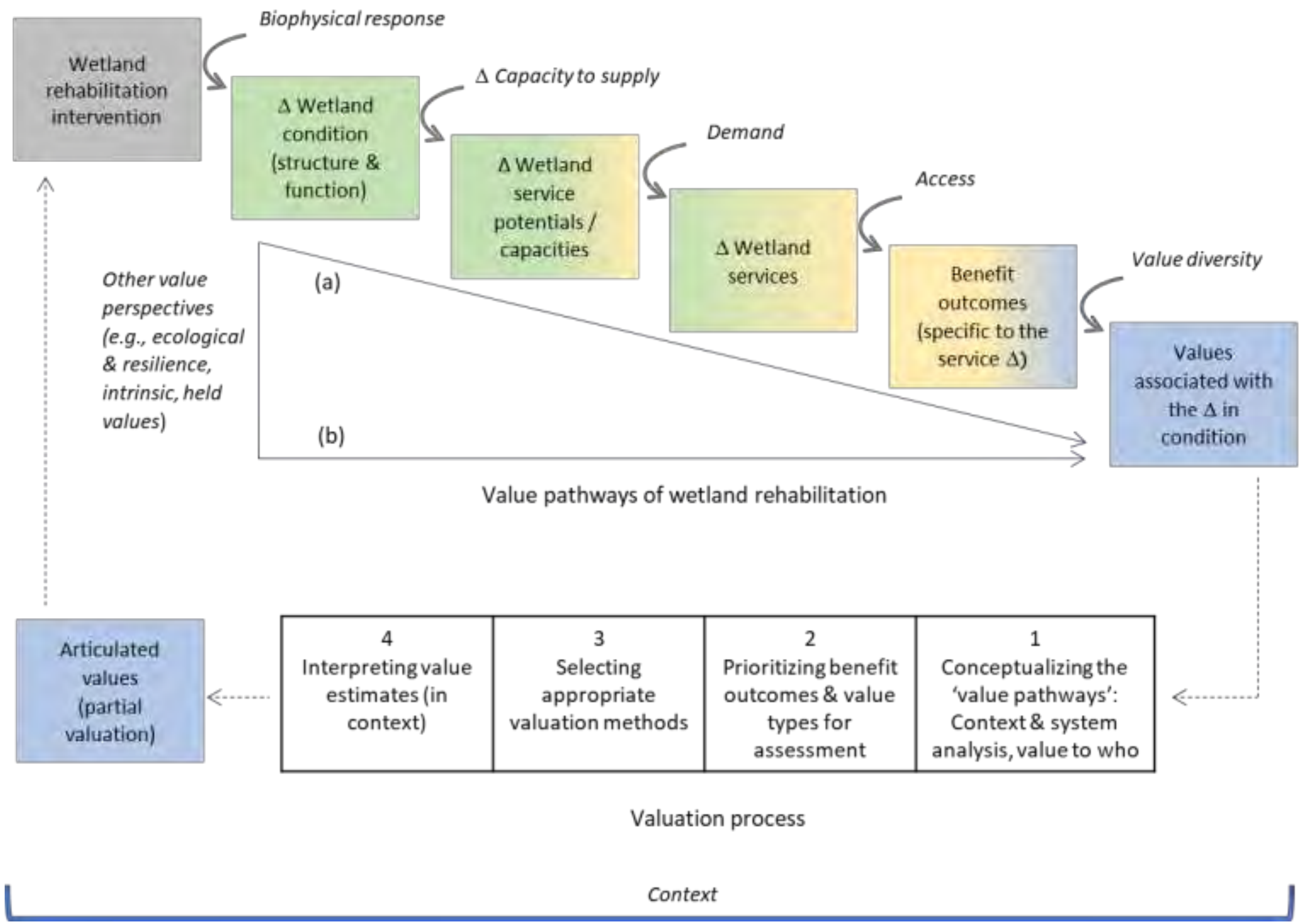


Figure 7.1: Guiding framework for the valuation of wetland rehabilitation.

In South Africa, wetland rehabilitation is intended to assist in the recovery of the condition or health of a degraded wetland or to halt the decline in condition of a wetland that is in the process of degrading (Kotze et al., 2008). The valuation of wetland rehabilitation is, therefore, concerned with the values associated with a change in wetland condition, in contrast to the values of the 'whole' system. The philosophical perspective of environmental value, along with multiple value classifications and conceptual metaphors of human–nature relationships (e.g., Raymond et al., 2013; Arias-Arévalo et al., 2018; Chan et al., 2018; Muradian and Pascual, 2018 and the TEV framing, Chapter 2) attest to a diversity of potential ecosystem values, which cannot be reduced to a single type or expressed through a single valuation method or approach (Martinez-Alier et al., 1998; Light, 2002; Norton and Noonan, 2007; Arias-Arévalo et al., 2018). It follows that well-defined differences in wetland condition between the 'with' and 'without' rehabilitation alternatives, based on wetland science, and a value pluralism perspective are fundamental to the valuation of wetland rehabilitation.

The ecosystem services concept, which has been adopted in the field of wetland management in South Africa, provides one framework for connecting ecosystem structure and function to human well-being. There are various interpretations of, and nuances to, the ecosystem services concept. In the proposed valuation framework, the Ecosystem Service Cascade introduced by Haines-Young and Potschin (2010)[102] has been adapted to the context of ecosystem change to depict one 'pathway' of wetland rehabilitation value articulation (pathway (a), Figure 7.1). The cascade model has been further modified to emphasize particular elements, identified through the research, relevant to wetland rehabilitation valuation.

Drawing from the theory, and highlighted in the case study applications, pathway (a) of the valuation framework distinguishes between the capacity of a wetland to supply a service (wetland service potentials in Figure 7.1) and the degree of realization of that capacity as a service, which depends on the appreciation of, and demand for, the additional capacity. A

[102] The cascade model has seen several iterations by the same group of authors (e.g., Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011) drawing from other frameworks and conceptual definitions (e.g., de Groot et al., 2002 and Boyd and Banzhaf, 2007), as well as modifications by other authors (e.g., Spangenberg et al., 2014a,b; Nassl and Löffler, 2015).

difference in wetland service reflects the actually utilized or desired quantity or quality of ecological phenomena, whereas 'potentials' reflect the capacity or capability of the wetland to provide an output that is potentially of benefit to people (after Fisher et al., 2009; Haines-Young and Potschin; 2010; Bastian et al., 2012). Conflating service supply capacity with the actual provision of wetland services risks over estimating the value of the rehabilitation. Arguably, however, there is some level or form of value attached to service potentials as in option, bequest and risk mitigation values.

A distinction is drawn between ecosystem service and benefits to emphasize that benefits derive from ecological phenomena in combination with additional inputs and enabling conditions (Ribot and Peluso, 2003; Wainger and Manzotta, 2011; Turner et al., 2016). 'Factors of access' – factors affecting the ability to derive benefit from an ecosystem service potential (e.g., inputs in the form of other 'capitals', legal rights, socio-cultural norms, demographic characteristics) - mediate the benefits which are, or can be, realized from wetland services and how benefits are distributed across different social groups and individuals. The communal land-tenure arrangements of the Manalana and Xharas wetlands allow people to derive benefit from the wetland in terms of consumptive use; the private land-tenure context of the Edendale Mall, on the other hand, limits such benefits.

A final distinction is made between benefit outcomes and values. In the proposed framework, 'values' are interpreted as the multiple, and not necessarily commensurable, ways in which wetland properties, services and associated benefits are important to individuals and social groups. The multiple interpretations of, and debates about, environmental values and ecosystem valuation evident across various schools of thought and fields of practice (Chapter 2) serve as a strong argument for value pluralism in the sense of multiple types of value, which are not necessarily substitutable or comparable in terms of being reduced to a single measure or expression. This was evident in the case studies where various types of values emerged including different dimensions of the TEV framework as well as ecological values (e.g., Xharas and Edendale Mall cases), relational values (e.g., Manalana and Baynespruit cases) and fundamental values (e.g., food security and livelihood support values evident in both the Manalana and Baynespruit cases). Conflating 'benefit' with 'value' fails to recognize the diversity of values that may be associated with a single type of benefit by the same individual or across different individuals. A caveat in this respect is the risk of 'double-counting' in that measured or elicited values may encompass several value types or dimensions. This aspect is taken-up further in section 7.1.2.3.

The modified ecosystem services framing linking the biophysical outcomes of wetland rehabilitation to values is illustrated as 'value pathway (a)' in Figure 7.1). There are also other ways of thinking about the relationships between people and wetlands and other types of value concepts that do not necessarily 'fit' within the ecosystem services framing. This is reflected as 'pathway (b)' in Figure 7.1. In contrast to the ecosystem framing, these value types may be associated with a change in wetland condition without recourse to demonstrating a direct link to a single specific benefit to people (e.g., intrinsic values, planetary systems maintenance and resilience contributions). There are considerable complexities in the causal links between changes in wetland condition and values; while the valuation pathways provide a means of conceptualizing relationships, they may mask the practical reality of the complexity of defining and measuring these interactions.

Implicit in thinking about the potential values of rehabilitation is considering 'value to who', with particular attention to identifying marginalised individuals or groups who's interactions with, and level of dependence on, the wetland and its associated functions may not be immediately obvious to outsiders and who may not have the 'power' to make their views heard. Power relations influence the ability of people to derive benefits from wetland systems and the distribution of benefits across different groups. Socio-ecological relations thinking contends that power (formal and informal) produces the relationships between society and non-human nature (Linton and Budds, 2014).

As emphasized in the theory and experienced in the case studies, ecosystem values are context dependent and specific to a particular point in time. An understanding of the interlinked ecological and social context is fundamental to identifying and measuring the values of wetland rehabilitation. Social-ecological systems and social-ecological relations thinking both emphasise the interconnections between human and non-human elements; it is through these relationships that rehabilitation values originate. Wetland systems are intricately connected to the surrounding landscape and the benefits resulting from wetland structure and function may be realized at a local, catchment, or even global scale. This is emphasized in what Hein et al. (2006) refer to as 'institutional scale' which reflects beneficiaries from the level of the individual and household through to national and international scales. The importance of a holistic understanding of the context and the interactions between biophysical and social elements highlights the interdisciplinary and even transdisciplinary nature of wetland rehabilitation valuation and the need for collaborative efforts across disciplinary experts and the participation of a range of relevant stakeholders.

The proposed framework further outlines four components (boxes 1 to 4, Figure 7.1) that emerged from the research as particularly important in the process of valuation. These are discussed in the following sub-sections.

7.1.2.1 Conceptualizing the value pathways

Wetlands are multifunctional complex systems, meaning that rehabilitation efforts can generate or enhance a wide range of benefits, but also that these outcomes can be challenging to predict and quantify precisely. Added to this, are the numerous ways people interact with, perceive and 'value' wetland systems and their contributions to people; these diverse interactions and values, often interlinked, are not always easy to define and simple to quantify. Practical considerations and theoretical challenges or limitations mean that value assessments are generally restricted to a subset of benefit outcomes and value types more possible and practical to quantify. Arguably, this is further influenced by a still common desire to measure values in monetary terms. This was evident in the studies reviewed in Chapter 3 and in the research case studies, which all tended to demonstrate partial valuations. Yet, this fails to make apparent the diversity of potential values, many of which may not be practical or possible to quantify, with potential implications for wetland management and rehabilitation decision-making. Articulating only a sub-set of benefits and / or values, and, similarly, favouring a single valuation approach, can obscure other types of value and other 'beneficiaries' and exclude or 'crowd out' other motivations for investing in wetland rehabilitation. The case study experiences further showed that 'values' emerged during or through the valuation process itself, specifically through the context analysis and multi-stakeholder engagement processes. Defining the assessment scope and selecting suitable methods prior to conceptualising the range of potential outcomes and value types could lead to a situation where relevant benefits, beneficiaries and values remain hidden.

In the Manalana case, for example, the monetary-based valuation of the direct consumptive use values of three wetland resources clearly underestimated the value of the rehabilitation in not accounting for food security, relational and option values. This had implications for the cost-benefit analysis, which, based only on market-based direct use values, indicated that the rehabilitation may not be worthwhile. Interpreting the direct use values in the context of the 'missing' values, however, provided considerable justification for the investment in the rehabilitation. These 'additional' values became apparent during the process of the assessment at which point the market-based direct use valuation approach had already been selected. A more open-ended conceptualization process, including the focus group discussions, undertaken before method selection and data collection design, could have enhanced the study by informing a wider selection of methods (including non-monetary based valuations) to more holistically assess these other aspects which are also part of the contribution of the rehabilitation to local livelihoods, the primary aim of the rehabilitation.

Both the theory on ecosystem valuation and the case studies attest to the importance of developing an understanding of, or familiarity with, the context of the rehabilitation, across the physical (ecological, technical) and societal (social, individual, economic, political) elements of the system and their interactions. This understanding is integral in considering the supply, demand and access aspects of the valuation. Context analysis approaches such as the STEEP (Social, Technical, Economic, Environmental, Political) framings and adaptations thereof have been applied within the environmental management field (e.g., Pollard et al., 2014; Bowd et al., 2015; Everard, 2015; Everard et al., 2021)[103] to develop a sense of the system and interactions between its elements. Gomez-Baggethun et al. (2014) suggested, however, that a comprehensive context analysis process is under-applied in the practice of ecosystem valuation.

[103] The VSTEEP framework, modified specifically to emphasise the 'value' context, has been applied in South Africa, often at a local scale (Pollard et al., 2014). The Social-Ecological System Analysis framework (Ostrom, 2007, 2009) and the South African adaptation of Bowd et al. (2015) and are relatively more analytically detailed approaches. The Social-Ecological Systems analysis framework of Ostrom (2007, 2009) is an extensive nested, multi-tier hierarchy of suggested variables for analysing social-ecological systems.

To this end, a component of conceptualizing the value pathways (Box 1, Figure 7.1), involving an analysis of the context[104], has been introduced into the framework to emphasise the importance of context and the identification of the range of human-wetland-rehabilitation interactions and potential values associated with the rehabilitation, and implicit, 'value to who', even if all potential values won't be measured or assessed in detail. To conceptualize is to form an idea about a particular system - its primary components and the relationships between them, for a particular purpose (Fischenich, 2008). In this way, potential benefits and values can be identified and factors affecting supply, demand and access revealed. Method selection may then be better informed to achieve a more holistic perspective and those values that are assessed in detail can be interpreted, or situated, within a wider recognition of potential values and what, and who, has been 'missed' out. This further assists in gauging whether the 'value' of the rehabilitation has been significantly underestimated or if a particular group of people or type of value is underrepresented by the measured values.

Further to being an essential part of any valuation, an analysis of the context and system conceptualisation provides valuable insights on the role and importance of the wetland and the contribution of the rehabilitation without recourse necessarily to value measurement. This is particularly noteworthy for rehabilitation evaluation cases where resources (time and budget) and data may be particularly limited, or where demonstrating the links between the rehabilitation outcomes and human well-being is sufficient with respect to management or decision-making information needs. Indeed, insights gained thorough a conceptualization process may reveal additional questions that need to be addressed prior to wetland rehabilitation valuation assessment. This was the case for the Baynespruit Stream study; consensus from the multi-stakeholder workshop suggested that wetland rehabilitation was not necessarily the most feasible or priority intervention for achieving the priority objectives and that alternatives should be considered. The insights also revealed a lack of ecological knowledge of the wetlands being considered and, therefore, limited information of likely outcomes of the rehabilitation on which to base a valuation assessment.

[104] During the course of this research, a number of contextual aspects relevant to wetland rehabilitation valuation were identified; Appendix 7.2 introduces several examples of these contextual factors.

7.1.2.2 *Prioritizing benefit outcomes and values for detailed assessment*

As was the case for the research applications, and apparent in the valuation studies reviewed (Chapter 3), detailed valuation assessments are generally limited to a sub-set of benefit outcomes and associated values given resource constraints. Several aspects can influence assessment priorities. The primary consideration being the aims of the rehabilitation and the expected outcomes. In many cases, wetland rehabilitation is designed with a specific aim in mind, as for the Mthinzima (improve water quality enhancement services), Edendale Mall (maintain no-nett loss of habitat) and Manalana (secure livelihood benefits) wetland rehabilitations. In such cases it is arguably not pragmatic, or even fair, to evaluate the rehabilitation based on outcomes and values for which it was not specifically intended. For example, evaluating the Edendale Mall wetland based on a contribution to recreation or natural resource harvesting would arguably not be fair given the primary aim to off-set wetland habitat loss as a result of the mall development and, further, the private ownership context of the wetland.

However, the conceptualisation process may reveal other outcomes and associated values not necessarily identified during the planning of the rehabilitation. Pre-rehabilitation, the conceptualisation process could highlight potential trade-offs or conflicts across wetland uses, affected people and value types. For example, trade-offs between the 'use' of the wetland for water quality improvement versus crop cultivation, or direct consumptive use values versus ecological values or sustainability concerns which may indicate the need for additional interventions such as the provision of irrigation to support crop cultivation away from the wetland, or the introduction of sustainable wetland use practices. A caveat for valuation being that the values associated with incompatible wetland 'uses' cannot be aggregated; however, a valuation assessment could be used to compare the values associated with conflicting uses.

Such insights could also influence design of the rehabilitation. In the Mthinzima case, for example, the water quality improvement imperative was arguably 'enough' to justify the rehabilitation, but the rehabilitation could also affect local livestock grazers in respect of access to water for the animals. To this end, the rehabilitation was designed to be both resilient to livestock impacts, but also to create points of safe access for livestock to drink. This example further highlights that prioritization of values for detailed assessment may not necessarily be based on the 'greatest' value created, but may be about the 'value to who' and a need to better understand the values held by smaller or marginalised groups. In the Mthinzima case, the value associated with the primary objective of the rehabilitation is arguably apparent in the number of downstream users, the importance of the water supply system and the risk to the system of further pollution inputs and, as such, may not need to be quantified to justify the investment. A better understanding of the consumptive use values related to livestock grazing and, further, the community relationships and social cohesion elements attached to these practices, on the other hand, may be needed to inform or justify a particular design of the rehabilitation to accommodate multiple objectives.

In the case of post-rehabilitation valuation, in addition to being guided by the rehabilitation aims, more detailed assessment of other outcomes and values may be warranted. For example, demonstrating co-benefits and values generated through the rehabilitation can provide additional justification for the investment and inform and further motivate future investment. Furthermore, additional outcomes and values identified post-rehabilitation may be particularly relevant from an adaptive management perspective. In the Manalana case, six years post-rehabilitation, the wetland was still making a significant contribution to local livelihoods with tentative indications of the preference of local residents for this option to be available for future use and to future generations. At the same time, the context of the village remains one of socio-economic vulnerability. Yet the ecological setting suggests that further investment in the maintenance of the existing rehabilitation structures and likely additional structures will be required in future to continue to stabilize the system. As such, greater exploration of option value and likely future use and demand could assist in informing whether further investment is warranted.

The intended use of the valuation information can also influence the prioritization of outcomes and values for more detailed assessment. If, for example, the intention is to raise awareness with the aim of securing funding from donor groups or international aid organisations then the valuation could focus on those outcomes and values that resonate with the target funders(s), even if they are not necessarily the primary aim of the rehabilitation. For example, valuation of the additional carbon storage service as a result of rehabilitation could be used to motivate for support from climate change funds. Another example might be valuation with the intention of obtaining consent from a landowner for wetland rehabilitation, undertaken through a state programme, and commitment from the landowner for long-term maintenance of the rehabilitated wetland. At a strategic, or programme level, valuation could be used to compare a number of potential rehabilitation projects with the intention of selecting between options based on a pre-defined set of key benefit outcome and value objectives. Or, post-rehabilitation, to build a base of primary valuation assessments of a particular benefit or value type to better understand under which conditions particular outcomes are achieved and / or for interrogating the potential use of primary valuations in predictive and value transfer approaches as practical tools to inform decision-making.

7.1.2.3 Selecting appropriate valuation methods

In the proposed framework, method selection (Box 3, Figure 7.1) has been purposefully placed after identifying and prioritizing likely benefits and values and developing an understanding of the context. Theory emphasizes that the valuation method applied must be consistent with the concept of value being considered (Bockstael et al., 2000; Fisher et al., 2008; Chan and Satterfield, 2020). In other words, that the value theory underlying the method aligns with the concept of value being considered and that the principles and assumptions of the value theory hold in the context of the application. For example, economic valuation methods based on a theory of exchange value, marginal individual utility and rational choice are not suited to unique goods[105] and cases of non-marginal changes and levels or thresholds of critical natural capital and do not adequately reflect many relational and social values. This suggests that an understanding of, or interrogation of, the theory underlying different value concepts and the associated valuation methods is needed. In other words, economic valuation methods should be applied with care. This was apparent in the

[105] The interpretation of 'unique goods', in an ecosystem context, is contested and relates to whether a weak or strong sustainability perspective is adopted.

application of the replacement cost method in the Mthinzima case study, which questioned whether the conditions for reliable application were met.

This thesis has specifically explored economic value theory and drawn attention to its limitations, particularly of neoclassical theory, in respect of holistic ecosystem valuation, and highlighted the assumptions and conditions which underly economic valuation methods. The case of wetland rehabilitation, and the characteristics of wetland services and benefits, pose a number of challenges to these assumptions and, therefore, to how reliable economic value estimates are when these assumptions or conditions are uncertain or do not hold. This implies that attention to several aspects and conditions that can affect the reliability of economic valuation, such as the 'economic good' properties of the benefit outcome, whether marginal conditions hold, and whether well-defined differences between the 'with' and 'without' rehabilitation case can be measured, is needed in considering whether to apply economic valuation methods[106].

In this respect, scarcity can be a perplexing and conflicting 'property' to unravel – *scarce at what scale, for who, in what timeframe, are there 'substitutes' and do they provide the same 'utility' and have the same values?* The standard economic interpretation of scarcity is a relative property, the interaction of supply and demand, yet scarcity is also mediated by mechanisms of access and control. Theoretically, standard economic valuation is not suited to 'goods' that are, or are perceived to be, abundant. In this way, there is a paradox in the argument of using economic valuation to motivate or justify investment in wetland rehabilitation when the purpose of the rehabilitation may be to prevent 'goods' from becoming 'scarce'[107]. Views on ecosystem and natural capital 'stock', critical natural capital, weak and strong sustainability all have relevance in interpreting 'scarcity'.

[106] These aspects or factors are presented as a summary table Appendix 7.3 which could be further developed an incorporated into an application manual or step-by-step guide in support of the valuation framework.

[107] In other words, there is a potential paradox in employing a concept of value that is intricately reliant on a condition of scarcity to provide motivation for maintaining or enhancing 'abundance'. In addition, there are arguments suggesting that attributing exchange value to the contributions of ecosystems risks 'commodifying' them and, essentially, privatising public wealth. Arguments along these lines echo the 'Lauderdale paradox' and compel reflection on the 'value in use' and 'value in exchange' concepts (Chapter 2).

In the Xharas case, the greatest contribution of the rehabilitation was to the maintenance of biodiversity. This was in terms of the wetland's location both in an area where wetlands have been identified as special habitats in need of conservation attention and within a global biodiversity hotspot, a centre for plant endemism and an important high water-yield area. Within the local landscape rehabilitation of the Xharas wetland made a significant contribution in terms of decreasing cumulative loss of wetland habitat and enhancing landscape connectivity, contributing to hydrological connectivity, linking previously fragmented patches of wetland as well as being part of a larger network of wetlands within the catchment. These contributions are ecologically important, both in their contribution to biodiversity maintenance and also in the context of improved system resilience, particularly important in respect of predicted climate change effects. Turner et al. (2008) described this role of wetlands as having 'contributory value', which emphasizes the complex relationships of interaction and interdependence between elements and species within ecosystems and, therefore, the limited substitutability within the web of biodiversity. Economic valuation is not suited to expressing the value of these ecological and system resilience contributions of the Xharas wetland rehabilitation, given the uniqueness of the system, the context of a system possibly near threshold levels, and, therefore, non-marginal changes, and the indirect, and therefore difficult to define and quantify, links between the biophysical outcomes and human well-being changes. In this case, ecological value indicators such as changes in wetland extent and integrity and measures of resilience are more appropriate. De Groot et al. (2010:23) emphasized that ecosystem integrity and resilience, as measures of value should be "distinguished from what can be included in economic values because although they contribute to welfare, they cannot readily be taken into account in the expression of individual preferences, as they are too indirect and complex, albeit they may be critical for human survival".

The Xharas case further highlights the cumulative contribution of multiple intact wetlands. Many of the services of wetlands, particularly regulatory services, but also biodiversity maintenance, have a greater impact at a catchment scale from the cumulative contribution of multiple intact wetland systems. The additional contribution, and value, of a single wetland rehabilitation may appear relatively small considered alone, while much greater when considered as part of the impact of multiple functional wetland systems within a catchment. Paradoxically, the scarcer the wetland service, the greater it's potential economic value.

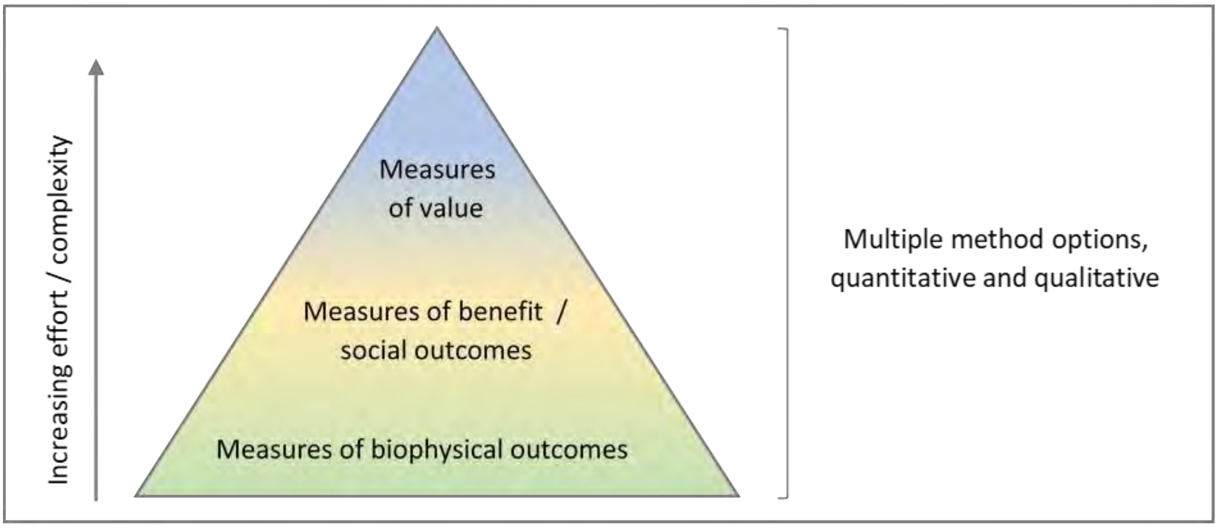
Stated preference methods are theoretically more expansive in the types of value that could be assessed or measured and can, for example, elicit both individual and social values (Hansjürgens et al., 2017) and multiple dimensions of the TEV framing, and it has been suggested that choice modelling can be used to explore relational values (Schulz and Ortega, 2018). This can also be problematic however, in that it may be unclear, or difficult to separate out, different sources or types of value when using stated preference methods. Depending on the context of the evaluation and information needs, this may not be an issue pragmatically, but 'double-counting' is a risk when applying multiple valuation methods. Kenter et al. (2015), in exploring the relationship between the TEV dimensions and 'shared' and 'social' values, alluded to the fact that even within a TEV-based framing value types can 'over-lap' and different methods are not mutually exclusive in the elicitation or expression of values.

Valuation methods are not ideologically neutral, having been developed from a particular understanding of what values are, or should be, and how they can be known, meaning that they impose a specific way of thinking and shape and define the values that are elicited (Vatn and Bromley, 1994; Martín-López et al., 2014). In the words of Gomez-Baggethun et al. (2014:20), "[v]aluation methods and associated rationalities are frames invoked in the process of expressing values that regulate and influence which values come forward, which are excluded, and what sort of conclusions can be reached". The implication for wetland rehabilitation valuation being that adopting any single valuation method will only result in a partial valuation and risks displacing or obscuring other values, forms of knowledge and human-wetland interactions. Similarly, the value metric adopted also influences the type of value expressed. This highlights the importance of understanding the context and identifying the range of potential values before introducing valuation methods, and interpreting measured values in the context of 'missing' values. It further suggests that a combination of methods are likely to be needed in any valuation assessment.

The limitations, or danger even, of relying on a single valuation method can be illustrated using the Manalana wetland case as an example where market-based value estimates of consumptive use alone, do not adequately portray the food security importance of the wetland and, specifically, the lack of alternatives to the dietary variety provided through wetland cropping. The risk being, especially given the weak CBA results, that decision-makers could favour compensation in the form of a monetary equivalent as an alternative to rehabilitation. Such compensation, based on the estimated values, would be an insufficient substitute to the wetland in terms of food security given the lack of local alternative food sources of the same variety. Furthermore, the prevailing patriarchal culture makes it likely that the men of the household would have a 'right' to at least a portion of any monetary compensation, leaving less for food provision, typically the role of woman.

The context of the beneficiaries or 'value holders' is a further consideration in selecting methods. This is emphasised in the debates on stated-preference valuation techniques, such as arguments raised in a developing economy context where "public understanding and knowledge of biodiversity concepts and relationships as they are used in the scientific literature are often low" (Christie et al., 2012:74) and "monetary economies are weak, where there is strong dependence on biodiversity, and where collective decision-making is common" (Kenter et al., 2011: 507). The links between wetlands and human well-being, and the influence of rehabilitation efforts on these connections, are complex and often indirect (less tangible), meaning that many people may not recognize the benefits of wetland rehabilitation and are unlikely to have pre-determined preferences in the respect of wetland rehabilitation. On the other hand, they may have well-defined preferences regarding specific benefits (e.g., cleaner water) without necessarily recognizing the connection between rehabilitation efforts and the availability of the benefit. Combining deliberative and participatory processes with stated preference techniques has been proposed as a means to address some of these challenges (e.g., Spash, 2008).

Method selection, with respect to wetland rehabilitation valuation, entails recognizing that multiple valuation methods are likely necessary, or at least that a single method will yield a partial valuation, that both biophysical and social information are fundamental, and that supporting processes or methods may be needed to facilitate valuation. This implies that method selection then, rather than being about choosing the most appropriate valuation method, is about selecting the group of methods, which, together, are best suited to expressing the outcomes and values identified as assessment priorities. These methods and different forms of information can be viewed from a 'tiered' perspective, with valuation measures as the top 'tier', as illustrated in Figure 7.2, supported by biophysical and social information and outcome measures. These biophysical and social outcomes measures may themselves be indicators of value, and, depending on the rehabilitation goals and management information needs, value measurement is not always be necessary.



Note: A pyramid has been used to emphasize that biophysical information underpins benefit and value measures, but also that not all biophysical outcomes have direct benefit outcomes and similarly that not all benefit outcomes can be assigned a measure or level of value (hence the placement of measures of value in the narrowest section of the pyramid). The merging of colours in the figure indicates that the boundaries between these groupings are permeable.

Figure 7.2: Conceptual illustration emphasising that multiple measures, and methods, are needed in wetland rehabilitation valuation.

There are several additional reasons for choosing between valuation methods. The case study applications highlighted the availability of biophysical data on which to base the valuation as a key determinant. In the Edendale Mall case, monitoring data, while adequate to demonstrate the positive influence of the rehabilitation on water quality, was insufficient to support economic valuation in that only water quality concentrations were measured and not water flows, meaning the actual quantity of pollutants removed or trapped by the wetland could not be calculated. In this case, however, the biophysical assessments demonstrating the ecological value of the rehabilitation were sufficient to indicate achievement of the rehabilitation objectives. In the Mthinzima case, constraints to hydrological modelling meant that an indirect valuation method was adopted. In the Baynespruit case, a lack of knowledge of the wetlands in the system and the likely outcomes of the rehabilitation meant that the 'with' and 'without' rehabilitation cases on which to base the valuation, could not be formulated, but also that it was not yet clear what could be achieved through rehabilitation. In this case, a much broader multi-criteria assessment approach across a range of options was a more appropriate starting point than rehabilitation valuation.

Valuation methods and approaches have different levels of data, time, budget and expertise requirements, which can influence the choice of methods. Using the Baynespruit stream case as an example, applying hedonic pricing methods to value improved storm water attenuation through wetland rehabilitation would require more detailed data than applying a damage cost avoided method. Arguably though, the two methods measure different aspects. In the Xharas case, the broad scope of the valuation and resource constraints meant that a more rapid, less resource intensive approach, able to give an indication of the contribution and importance across a wide range of benefits and values was needed, even at the expense of some level of assessment rigour and certainty or confidence in the results. Whether the results derived using indicator or indirect approaches (e.g., cost-based methods or expert judgement-based index approaches) are considered acceptable depends on the intended use of the valuation information; certainty and confidence requirements between awareness raising and compensation determining uses, for example, will be different.

The intended use of the valuation information can influence method selection for other reasons. For example, if valuation is intended to motivate for funding from particular groups, then methods that generate information in a form that is relevant and accessible to the intended audience might be favoured (e.g., if the target group is a birding community or organisation then ecological value measurement methods that articulate the relationship between the rehabilitation and suitable habitat and bird prevalence and diversity would be best suited). Similarly, there may be objectives associated with the valuation process itself such as fostering the building of a common understanding across diverse stakeholders. In such cases, tools that facilitate stakeholder participation and encourage dialogue and deliberation would be appropriate.

7.1.2.4 Interpreting valuation estimates

'Interpreting value estimates' (Box 4, Figure 7.1) has been specifically included in the valuation framework to emphasise the importance of interpreting the results in terms of what has been measured and what has not, what do the results mean for the specific beneficiaries involved, and what are the uncertainties. This research has suggested three key aspects to the interpretation of valuation findings

A first aspect, as implied by the partial nature of value measurement, is the interpretation of value estimates with respect to the coverage of the potential values identified and which values and benefits are missing from the detailed assessment and, importantly, whose values are missing and potential distributional issues, and the likely implications for conclusions regarding the rehabilitation objectives and the overall value of the rehabilitation. Returning to the Manalana case study, food security and safety net values were not measured directly, but were revealed through the valuation process as key sources of value. Similarly, option and bequest values emerged as important dimensions not investigated directly. Applying only monetary valuation of direct use values in this case, undervalued the role and contribution of the rehabilitation. The case study also emphasised the need for attention to the context of the value holders or beneficiaries. Considering the market value of additional crop production in terms of the average household income of the farmers was essential to understanding the importance of this contribution (as both cash income and spending avoided) even if the monetary value estimates appear relatively low. In the Mthinzima case, the downstream recreation and concomitant economic implications of water quality changes were not reflected in the assessment; while less of a priority than potable water implications, the downstream water supply dam is highly used for recreation, including national sporting events. Typically, valuation assessments will only measure or assess in detail a portion of potential values and benefits and the perspectives of a sub-set of affected groups; this was evident in the review of wetland rehabilitation studies (Chapter 3) and the case studies.

Without critical interpretation of the results reflecting on what has not been expressed in the value estimates, partial valuation risks misconstruing rather than revealing.

The field of ecological economics supports both value and methodological pluralism and takes the view that ecosystem valuation entails taking account of a range of value types, which may not be commesurable to a single metric. The approaches and methods developed within Ecological Economics offer alternatives to classical project appraisal methods, whereby different value types may be integrated to inform decisions without being reduced to a single measure. Such methods can be drawn on in operationalising the proposed framework, specifically in terms of combining and comparing values in the interpretation of benefit outcomes.

A second aspect of interpretation is a reflection on the sources and extent of uncertainty and the strengths and limitations of the methods applied. The research highlighted uncertainties stemming from both the understanding and measurement of biophysical changes and the application of the valuation methods. Clearly communicating these uncertainties and assessment limitations is important to ensure that low confidence information is not overly relied on in decision-making and the results inadvertently misused. Without a reflection on the uncertainties and the limitations of the assessment, valuation can simulate certainty where this might not be the case. An indication of the level of uncertainty could be enhanced through reporting error margin analysis results for biophysical measurements as recommended by Spangenberg and Settele (2016); using scenario analysis to generate value estimate ranges; and, for cost-benefit analysis, sensitivity analysis of key parameters. Without a reflection on elements of uncertainty and level of confidence, valuation results could imply certainty where this might not be the case. Whether the level of uncertainty or confidence is considered acceptable would depend on the intended use of the valuation information.

While perhaps more commonly associated with academic or research studies, a reflection on the assessment design and methods applied is more broadly warranted in the current context of the emerging nature of the field of wetland rehabilitation valuation to inform and improve practice going forward. This is particularly important in light of the complexity of wetland values and the variety of methods available, and continued development of new, and therefore not necessarily well-tested methods, as well as the uncertainties associated with predicting and quantifying cause and effect relationships between rehabilitation and social outcomes. Such a reflection could include attention to the ability of the methods to articulate the priority values (i.e., what was actually measured); limitations and challenges; contextual factors and supporting methods and information that improved the accuracy of, or confidence, in the results; and the influence of the methods themselves on shaping the type and level of values elicited. Furthermore, comprehensively reporting on the assessment design, methods applied and the specific ecological and social context of rehabilitation valuations can increase their utility for predictive valuation purposes, for exploring value transfer approaches and for use in meta-analysis.

A third aspect of interpretation pertains to the integration of the value estimates with other information. This could be in the context of a broader evaluation of the rehabilitation, bringing together findings across various components of evaluation (e.g., ecological effectiveness, sustainability, rehabilitation outputs, etc.) to address questions of how well the project addressed the original objectives (i.e., how the achieved outcomes compare with the intended outcomes) and the relevance of the strategy and objectives themselves or to inform adaptive management. Information integration could also be in the form of decision-analysis and synthesizing approaches such as cost-benefit analysis, multi-criteria analysis, Bayesian belief networks and scenario analysis to further inform decision-making, planning and prioritization, and motivate for funding.

7.2 CLOSING REMARKS

Recent global agendas and commitments to targets for ecosystem restoration, such as the Decade of Restoration 2021-2030 (UNEP and FAO, 2019), the Pan-African Action Agenda on Ecosystem Restoration 2019-2030 (Convention on Biological Diversity (CBD) and UNEP, 2018), and the rehabilitation targets as set as part of South Africa's commitments to Land Degradation Neutrality (DEA, 2018), suggest increasing pressure on governments to implement ecosystem rehabilitation and imply a concomitant increase in specific decision-making regarding where to rehabilitate, how best to design rehabilitation interventions to accommodate multiple objectives, and post-rehabilitation evaluation to demonstrate accountability to funders. The implementation plan for the Pan-African Action Agenda emphasises a need to understand the interactions between ecosystem restoration and people; the UN Decade of Restoration calls attention to considering how societies themselves perceive the value of ecosystem restoration. These information needs and local scale decisions require a more nuanced and context-orientated approach to valuation, compared to valuation for high-level awareness raising for instance; an approach that is able to articulate the range of outcomes and values for different people that may emerge, and so address more specifically the question of 'value to who'[108].

The neoclassical economic model, especially combined with a monetary metric, is too restrictive, and arguably too abstract in its assumptions of human behaviour and reliance on mathematical models, as an overarching framework for responding to these valuation needs. This is not to suggest that standard economic valuation concepts and methods cannot be useful, as the research case studies illustrated, but rather that valuation should not be approached from a single concept or dimension of value. An over reliance on any single value perspective, such as economic exchange value, can obscure or displace other types of values and human-wetland interactions and, therefore, potentially misinform specific decisions regarding wetland rehabilitation implementation and design while also influencing society's perception of value. While arguably this is the case for any value monistic perspective, given its pervasiveness and influence, an over reliance on an economic value framing has the potential to significantly shape humanity's perception of value and the human-nature relationship as warned by Maki (2011) among others. Norton and Noonan (2007) raised concerns with what they described as an enthusiasm for the ecosystem services concept used in conjunction with economic valuation methods, which has "locked the rhetoric of environmental evaluation in a very monistic, utilitarian, and economic vernacular that leaves little or no room for other social scientific methods, or for appeal to philosophical reasons or theological ideals" (Norton and Noonan, 2007:665).

[108] Such an approach compares with a 'bottom-up' assessment perspective, which attempts to draw out the nuances in a given location, often using a combination of quantitative and qualitative methods, and is suitable to including stakeholders and allowing them to voice their perspectives and objectives.

Furthermore, such an over-reliance means that where an economic value concept is not aligned with the goals of the rehabilitation, or where the challenges of estimating economic value are prohibitive, the values of the rehabilitation may remain unrecognized or articulated and ignored, arguably perpetuating the problem ecosystem valuation seeks to address. A further risk related to the pervasiveness of economic valuation is the adoption, or promotion, of standard valuation methods, commonly combined with a monetary metric, by those not familiar with their assumptions and limitations, leading to their use, or interpretation, in ways many economists would not. The potential of this occurring is suggested in the findings of the review on the economic valuation of wetland rehabilitation (Chapter 3) where almost half of the studies appear not to have included an economic department or institution[109]. This suggests a need to more widely communicate the scope and constraints of standard economic valuation methods and to bring a value pluralism perspective more strongly into the practice of ecosystem restoration valuation. That a value pluralism perspective is needed, is not a new idea in ecosystem valuation, but one, contend Arias-Arévalo et al. (2018), that remains to become common in application.

A value pluralism perspective has implications for how valuation, and research on ecosystem values, is approached. As observed in the research case studies, human-wetland interactions and values were revealed during the valuation process, or pre-conceptions about values and benefits challenged, specifically through engagement with affected people and various stakeholders. While this implies, as suggested in the valuation framework proposed by thesis, that method selection should only come after value identification, it also draws attention to how valuation, and valuation research, is approached. This is in respect of the influence of the positionality of the assessor or researcher and the role of those affected by the rehabilitation in the valuation process. Just as valuation methods are value articulating, those undertaking the valuation similarly shape and define the values that are elicited. This suggests that valuation, and research on wetland values, must be approached in a manner that is open to diverse human-wetland interactions and benefits and values that may emerge during the valuation. It further suggests a shift from viewing those affected by the rehabilitation as the

[109] However, this finding is based on author affiliation and would have excluded economists whose affiliation details did not indicate an economic connection.

'subjects' of valuation to one where they are active participants in the process, and raises the deeper philosophical questions of who has the right to assign value and who has the right to decide whose values, or which type of values, count more?

A related observation from the case studies was how the process of the valuation assessments generated discussion, debate and interaction across disciplines and stakeholders, which itself led to a deeper appreciation of the likely or observed contributions and disbenefits of the rehabilitation projects and the potential values, along with a better understanding across stakeholders of knowledge gaps and uncertainties in respect of value measurement. In this way the valuation process was meaningful, even if, in the end, valuation measurement was not possible or only possible for a few benefits and value types. These observations suggest that future work on wetland rehabilitation valuation explore valuation as a participatory, consensus seeking process in support of collaborative decision-making and the co-design of wetland rehabilitation goals and management strategies.

From a practical perspective, comprehensive outcomes valuation is unlikely to be feasible, or even necessary for every case of wetland rehabilitation; a rehabilitation intervention in a protected area with the aim of contributing to biodiversity maintenance for example. However, in cases where there are multiple potential benefits, beneficiaries and values and / or conflicting land-uses, a comprehensive valuation may be needed to assist decision-making and / or a post-rehabilitation valuation to guide adaptive management or demonstrate accountability. Furthermore, a sample of comprehensive post-rehabilitation valuation studies, supported by detailed monitoring, are needed to develop an 'evidence base' for pre-rehabilitation predictive studies and towards exploring rapid valuation approaches or indicators and whether there are some generalizable relationships between particular circumstances and likely value outcomes that could be used to guide broad-scale implementation. Rather than recommending comprehensive valuation as a necessary part of every project evaluation, detailed valuation could be used in a sub-set of cases across a range of contexts and locations, to build an understanding (and evidence base) of the values of wetland rehabilitation in a variety of South African contexts. It is suggested that such valuations be framed in a recognition of value pluralism and seek to explore the potential of valuation as a participatory, consensus-building process, and a wider range of valuation methods.

The question then arises of whether the economic discipline, or the 'economic mind' as phrased by de Wit (2019), can encompass a value pluralism perspective to provide an overarching framework for wetland rehabilitation valuation, or whether the scope of economics is confined to the dimension of economic exchange value and the associated value measurement methods. On the scope of economics, Coase (1978:208) wrote that "it by no means follows that an approach developed to explain behaviour in the economic system will be equally successful in the other social sciences" and argued that economic analysis will require major modifications to be successfully applied to social problems beyond the 'economic system'.

The transition to recognizing that economic, social and ecological systems are not separate systems and sustainability and equity as central social goals, suggests that a fundamental shift in human behaviour and decision-making is necessary to address current global challenges such as ecosystem degradation, climate change and growing inequality. Whether a fundamental shift in the philosophy of economics itself is needed, or whether the shift is more subtle in respect of how economic concepts and tools, are interpreted and applied, remains contested and closely related to the debate on weak versus strong sustainability. In a valuation context, questions include whether aspects such as fairness, to other people now and in the future and to non-human nature, moral duty, and shared values fall outside of an economic welfare framing. Is a desire to act fairly and sustainably outside of individual preference satisfaction? Is economic value synonymous with exchange value and substitution, as in the neoclassical economics interpretation, or is current economic thinking open to critical reflection, for example, on pre-neoclassical theories of 'use value' and 'wealth' (Foster and Clark, 2009; Gómez-Baggethun et al., 2010), on its relationship to ethics (Spash, 1999) and its reductionist assumptions (Maki, 2001)?

Indeed, much of the debate on the suitability of economic ecosystem valuation revolves around whether the economic efficiency criterion is appropriate and sufficient to guide decisions on public goods and human well-being in the context of sustainability and equity goals. As remarked by Marshall (1920: I, III) "every change in social conditions is likely to require a new development of economic doctrines". An expansion in social objectives, as evident in the adoption of the sustainable development goals, does imply the need for a new valuation language, one embedded in a value pluralism perspective. Certainly, the limitations of the neoclassical economics model are widely recognized and many have advocated for a meaningful transformation of the standard economic approach. While such shifts are apparent (Davis, 2006; Ekstedt and Fusari, 2010), in a recent reflection, de Wit (2019) argued that the effects of the 'the shifting mind of economics' are yet to become fully evident in economic policymaking.

This thesis has introduced several economic schools of thought that expand on, or diverge from, the strict neoclassical view, and provide broader insights on addressing the complex ways in which individuals and societies value 'goods'. Parks and Gowdy (2013:e6) argued however, that many of these ideas "have been relatively slow to influence environmental valuation techniques and policy analysis". Further work in this area has the potential to strengthen ecosystem valuation and may provide the platform for an expanded scope of economics in the development and application of ecosystem valuation. Jacobs et al. (2016:215) argued that the complexity of valuation and the challenges of practical application "defy hopes for a methodological silver bullet" and concluded that a combination of perspectives, disciplines and methods is needed to reflect the diverse values of nature. Ecosystem valuation is certainly an interdisciplinary undertaking.

How then is valuation to be 'taught', given its fundamentally interdisciplinary nature and value pluralism perspective[110]? Does economic training provide this knowledge diversity, and does it encourage students to engage critically with economic thinking and methodology and be open to learning? Such a 'posture' may be necessary for economists to "engage fruitfully with the shifting economic mind" (de Wit, 2019:258). Such questions warrant future reflection.

[110]In addition, mixed method (quantitative and qualitative) requirements.

In closing, two further comments are made. While wetland rehabilitation, and ecosystem rehabilitation more broadly, is a necessary activity it should not take priority over conserving and maintaining intact systems. Similarly, valuation should not focus only on rehabilitation per se, but also on the actions and investments required to conserve intact systems. Ecosystem maintenance is often less expensive than rehabilitation (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), 2018; Jewitt et al., 2020) and, importantly, ecosystem recovery is not immediate (Streever 1997; Moreno-Mateos et al. 2012) and rehabilitation does not 'restore' the pre-degradation condition. The importance of ecosystem conservation is formally recognized in the degradation response hierarchy (IPBES, 2018). An area of work for valuation then, is also to give attention to how best to articulate the value of maintaining intact systems. In this context, concepts of ecosystems as assets (Atkinson et al., 2012) and research on ecosystem asset valuation, such as that by Fenichel et al. (2014), merits attention. In this respect, a profound point is emphasised by Hettinger's argument that restoration should not be regarded as a net benefit to society, but rather as a compromise; "a short-term and fundamentally regrettable way of relating to nature" (Hettinger, 2012:41)[111].

There is much work still to be done when it comes to the valuation of wetland (ecosystem) restoration. However, ecosystem valuation is a much contested topic, partly because it exposes fundamental differences between the weak and strong sustainability perspectives.
In a valuation context, Freeman (2003) emphasised the connection between valuation and the underlying goals to which a society aspires, arguing that 'benefits' and 'costs' are meaningless unless set against a specific goal or objective. Ultimately, the development of ecosystem valuation is embedded in a much broader debate, one which is far from settled.

From a value pluralism perspective, the challenge remains of how to integrate and make sense of multiple values in a decision-making context. Valuation itself cannot answer the questions of whose and which type of values count more. A caution then, is a disproportionate focus on ecosystem value assessment, a consequence of which, as many

[111] These points raise questions for the notion of ecosystem off-sets, specifically in the case of damage to intact or healthy systems, and suggests the concept should be adopted with caution and careful regulation.

have argued, is a lack of attention to the institutional changes needed to address sustainability goals and the drivers of ecosystem degradation. By way of example, Groom and Turk (2021) suggest that valuation has been given a central place in the Dasgupta Review on the Economics of Biodiversity (2021), specifically valuation in terms of "pricing biodiversity properly" and the correction of markets, "when clearly other structural, institutional and political changes will be required" (Groom and Turk, 2021:19). Spash and Aslaksen (2015:252) proposed shifting the ecosystem valuation debate from "how best to convert ecology into economics towards what are the best institutions humanity can create that are able to articulate different values, empower silent voices and the disenfranchised, and recognise and address issues of injustice and abuse of power". Perhaps the broader shift in economic thinking taking place is an opportunity for economists to take up this challenge.

REFERENCES

ADEKOLA, O., MORARDET, S., DE GROOT, R. and GRELOT, F., 2012. Contribution of provisioning services of the Ga-Mampa wetland, South Africa, to local livelihoods. *International Journal of Biodiversity Science, Ecosystem Services & Management,* 8(3), pp. 248-264.

ADUSUMILLI, N., 2015. Valuation of ecosystem services from wetlands mitigation in the United States. *Land,* 4(1), pp. 182-196.

ALDRED, J., 1994. Existence value, welfare and altruism. *Environmental Values,* 3(4), pp. 381-402.

ALDRED, J., 2006. Incommensurability and monetary valuation. *Land Economics,* 82(2), pp. 141-161.

ALEXANDER, S. and MCINNES, R., 2012. *The benefits of wetland restoration.* Ramsar Scientific and Technical Briefing Note No. 4. Gland: Ramsar Convention Secretariat.

ALEXANDER, S., ARONSON, J., WHALEY, O. and LAMB, D., 2016. The relationship between ecological restoration and the ecosystem services concept. *Ecology and Society,* 21(1), pp. 34-43.

ALFRANCA, O., GARCÍA, J. and VARELA, H., 2011. Economic valuation of a created wetland fed with treated wastewater located in a peri-urban park in Catalonia, Spain. *Water Science & Technology,* 63(5), pp. 891-898.

ALLISON, S.K., 2007. You can't not choose: Embracing the role of choice in ecological restoration. *Restoration Ecology,* 15(4), pp. 601-605.

ANDERIES, J.M., JANSSEN, M.A. and OSTROM, E., 2004. A framework to analyze the robustness of social-ecological systems from an institutional perspective. *Ecology and Society,* 9(1), pp. 18.

ANDERSEN, T., CARSTENSEN, J., HERNÁNDEZ-GARCÍA, E. and DUARTE, C.M., 2009. Ecological thresholds and regime shifts: approaches to identification. *Trends in Ecology & Evolution,* 24(1), pp. 49-57.

ANGERMEIER, P.L. and KARR, J.R., 1994. Biological integrity versus biological diversity as policy directives: Protecting biotic resources. *Bioscience,* 44(10), pp. 690-697.

APFELBAUM, S.I. and HANEY, A., 2013. *Restoring Ecological Health to Your Land.* 1 edn. Washington, D.C.: Island Press.

ARIAS-ARÉVALO, P., GÓMEZ-BAGGETHUN, E., MARTÍN-LÓPEZ, B. and PÉREZ-RINCÓN, M., 2018. Widening the evaluative space for ecosystem services: A taxonomy of plural values and valuation methods. *Environmental Values,* 27, pp. 29-53.

ARMSTRONG, A., 2008. *WET-Legal: Wetland rehabilitation and the law in South Africa.* WRC Report TT 338/08. Pretoria: Water Research Commission.

ARONSON, J. and ALEXANDER, S., 2013. Ecosystem restoration is now a global priority: Time to roll up our sleeves. *Restoration Ecology,* 21(3), pp. 293-296.

ARONSON, J., BLIGNAUT, J.N., MILTON, S.J., et al., 2010. Are socioeconomic benefits of restoration adequately quantified? A meta-analysis of recent papers (2000–2008) in Restoration Ecology and 12 other scientific journals. *Restoration Ecology,* 18(2), pp. 143-154.

ARROW, K., DASGUPTA, P., GOULDER, L., et al., 2004. Are we consuming too much? *Journal of Economic Perspectives,* 18(3), pp. 147-172.

ARSEL, M. and BÜSCHER, B., 2012. Nature™ Inc.: Changes and continuities in neoliberal conservation and market-based environmental policy. *Development and Change*, 43(1), pp. 53-78.

ATKINSON, G., BATEMAN, I. and MOURATO, S., 2012. Recent advances in the valuation of ecosystem services and biodiversity. *Oxford Review of Economic Policy,* 28(1), pp. 22-47.

ATTFIELD, R., 1994. Rehabilitating nature and making nature habitable*. Royal Institute of Philosophy Supplement*. 36, pp. 45-57.

BACKHOUSE, R.E. and MEDEMA, S.G., 2009. Retrospectives: On the definition of economics. *Journal of Economic Perspectives,* 23(1), pp. 221-33.

BAIYEGUNHI, L.J.S. and MAKWANGUDZE, K.E., 2013. Home gardening and food security status of HIV/AIDS affected households in Mpophomeni, KwaZulu-Natal Province, South Africa. *Journal of Human Ecology,* 44(1), pp. 1-8.

BALMFORD, A., BRUNER, A., COOPER, P., et al., 2002. Economic Reasons for Conserving Wild Nature. *Science,* 297(5583), pp. 950-953.

BARBIER, E.B. and HEAL, G.M., 2006. Valuing ecosystem services. *The Economists' Voice, De Gruyter,* 3(3), pp. 1-6.

BARBIER, E.B., 1993. Sustainable use of wetlands valuing tropical wetland benefits: Economic methodologies and applications. *The Geographical Journal,* 159(1), pp. 22-32.

BARBIER, E.B., 2007. Valuing ecosystem services as productive inputs. *Economic Policy,* 22(49), pp. 178-229.

BARBIER, E.B., 2011. Wetlands as natural assets. *Hydrological Sciences Journal,* 56(8), pp. 1360-1373.

BARBIER, E.B., ACREMAN, M.C. and KNOWLER, D., 1997. *Economic valuation of wetlands: A guide for policy makers and planners.* Gland, Switzerland: Ramsar Convention Bureau.

BASSETT, T.J. and PEIMER, A.W., 2015. Political ecological perspectives on socioecological relations. *Natures Sciences Sociétés,* 23(2), pp. 157-165.

BASTIAN, O., HAASE, D. and GRUNEWALD, K., 2012. Ecosystem properties, potentials and services – The EPPS conceptual framework and an urban application example. *Ecological Indicators,* 21, pp. 7-16.

BATEMAN, I., MACE, G., FEZZI, C., ATKINSON, G. and TURNER, K., 2011. Economic analysis for ecosystem service assessments. *Environmental and Resource Economics,* 48(2), pp. 177-218.

BATIE, S.S., 2008. Wicked problems and applied economics. *American Journal of Agricultural Economics,* 90(5), pp. 1176-1191.

BAUM, S.D., 2012. Value typology in cost-benefit analysis. *Environmental Values,* 21(4), pp. 499-524.

BAUMGÄRTNER, S., BECKER, C., FRANK, K., MÜLLER, B. and QUAAS, M., 2008. Relating the philosophy and practice of ecological economics: The role of concepts, models, and case studies in inter- and transdisciplinary sustainability research. *Ecological Economics,* 67(3), pp. 384-393.

BAVEYE, P.C., BAVEYE, J. and GOWDY, J., 2013. Monetary valuation of ecosystem services: It matters to get the timeline right. *Ecological Economics,* 95 (2013), pp. 231–235.

BECKERMAN, W. and HEPBURN, C., 2007. Ethics of the discount rate in the Stern Review on the Economics of Climate Change. *World Economics,* 8(1), pp. 187-210.

BENDA, L.E., POFF, L.N., TAGUE, C., et al., 2002. How to avoid train wrecks when using science in environmental problem solving. *Bioscience,* 52, pp. 1127-1136.

BERKES, F., 2004. Rethinking community-based conservation. *Conservation Biology,* 18(3), pp. 621-630.

BHASKAR, R., FRANK, C., HØYER, K.G., NAESS, P. and PARKER, J., 2010. *Interdisciplinarity and climate change. Transforming knowledge and practice for our global future.* London: Routledge.

BINDER, C.R., HINKEL, J., BOTS, P.W.G. and PAHL-WOSTL, C., 2013. Comparison of frameworks for analyzing social-ecological systems. *Ecology and Society,* 18, pp. 26-45.

BIRCH, M.B.L., GRAMIG, B.M., MOOMAW, W.R., DOERING, I., O. and REELING, C.J., 2011. Why metrics matter: Evaluating policy choices for reactive nitrogen in the Chesapeake Bay Watershed. *Environmental Science and Technology,* 45(1), pp. 168-174.

BIROL, E., KAROUSAKIS, K. and KOUNDOURI, P., 2006. Using economic valuation techniques to inform water resources management: A survey and critical appraisal of available techniques and an application. *Science of the Total Environment,* 365(1–3), pp. 105-122.

BLACK, D., TURPIE, J., RAO, N., 2016. Evaluating the cost-effectiveness of ecosystem-based adaptation: Kamiesberg wetlands case study. *South African Journal of Economic and Management Sciences,* 19(5), pp. 702-713.

BLAUG, M., 1985. *Economic Theory in Retrospect.* Cambridge: Cambridge University Press.

BLIGNAUT, J., ARONSON, J. and DE WIT, M., 2014. The economics of restoration: looking back and leaping forward. *Annals of the New York Academy of Sciences,* 1322(1), pp. 35-47.

BLIGNAUT, J., ESLER, K.J., DE WIT, M.P., et al., 2013. Establishing the links between economic development and the restoration of natural capital. *Current Opinion in Environmental Sustainability,* 5(1), pp. 94-101.

BLIGNAUT, J.N., ARONSON, J., WOODWORTH, P., et al., 2007. Restoring natural capital: A reflection on ethics. In: J. ARONSON, S. MILTON and J.N. BLIGNAUT, eds, *Restoring Natural Capital: Science, Business, and Practice.* Washington, D.C.: Island Press, pp. 9-16.

BOCKSTAEL, N.E., KOPP, R.J., PORTNEY, P.R. and SMITH, V.K., 2000. On measuring economic values for nature. *Environmental Science & Technology,* 34(8), pp. 1384-1389.

BOTHMA, A.J., 2004. *Working for Wetlands Rehabilitation Plan for the Save the Sand Project 2005/2006.* Pretoria, South Africa: Working for Wetlands Programme, RSA National Government.

BOWD, R., QUINN, N.W. and KOTZE, D.C., 2015. Toward an analytical framework for understanding complex social-ecological systems when conducting environmental impact assessments in South Africa. *Ecology and Society,* 20(1), pp. 41-59.

BOXALL, P.C., ADAMOWICZ, W.L., SWAIT, J., WILLIAMS, M. and LOUVIERE, J., 1996. A comparison of stated preference methods for environmental valuation. *Ecological Economics,* 18(3), pp. 243-253.

BOYD, J. and BANZHAF, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics,* 63(2), pp. 616-626.

BOYLE, K.J., 2003. Contingent valuation in practice. In: P.A. CHAMP, K.J. BOYLE and C. BROWN THOMAS, eds, *A Primer on Nonmarket Valuation.* New York: Springer Netherlands, pp. 111-169.

BRAAT, L.C. and DE GROOT, R., 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services,* 1(1), pp. 4-15.

BRAND, F., 2009. Critical natural capital revisited: Ecological resilience and sustainable development.*Ecological Economics*, 68 (2009), pp. 605-612.

BRANDER, L., BRÄUER, I., GERDES, H., et al., 2012. Using meta-analysis and GIS for value transfer and scaling up: Valuing climate change induced losses of European wetlands. *Environmental and Resource Economics,* 52(3), pp. 395-413.

BRANDER, L.M., FLORAX, RAYMOND J. G. M. and VERMAAT, J.E., 2006. The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. *Environmental and Resource Economics,* 33(2), pp. 223-250.

BROEKX, S., SMETS, S., LIEKENS, I., BULCKAEN, D. and DE NOCKER, L., 2011. Designing a long-term flood risk management plan for the Scheldt estuary using a risk-based approach. *Natural Hazards,* 57(2), pp. 245-266.

BRONDÍZIO, E., GATZWEILER, F., KUMAR, M. and ZOGRAFTOS, C., 2010. Socio-cultural context of ecosystem and biodiversity valuation. In: P. KUMAR, ed, *The Economics of Ecosystems and Biodiversity (TEEB): The Ecological and Economic Foundations.* 1 edn. London: Routledge, pp. 149-174.

BROUWER, R. and GEORGIOU, S., 2012. Economic evaluation. In: A. DUFOUR, J. BARTRAM, R. BOS and V. GANNON, eds, *Animal Waste, Water Quality and Human Health.* London: IWA Publishing, pp. 429-459.

BROUWER, R., BRANDER, L., KUIK, O., PAPYRAKIS, E. and BATEMAN, I., 2013. *A Synthesis Of Approaches To Assess And Value Ecosystem Services in the EU in the Context of TEEB. TEEB Follow-Up Study for Europe, Final Report.* Amsterdam: University of Amsterdam.

BROWN, T.C., 2003. Introduction to stated preference methods. In: P.A. CHAMP, K.J. BOYLE and T.C. BROWN, eds, *A Primer on Nonmarket Valuation.* Dortdrecht: Springer, pp. 99-110.

BURKHARD, B., KROLL, F., NEDKOV, S. and MÜLLER, F., 2012. Mapping ecosystem service supply, demand and budgets. *Ecological Indicators,* 21, pp. 17-29.

BUSHBUCKRIDGE MUNICIPALITY, 2014. *Integrated Development Plan: 2014- 2016.* Bushbuckridge, South Africa: Bushbuckridge Municipality.

CABEZA GUTÉS, M., 1996. The concept of weak sustainability. *Ecological Economics,* 17(3), pp. 147-156.

CAFFEY, R.H., WANG, H. and PETROLIA, D.R., 2014. Trajectory economics: Assessing the flow of ecosystem services from coastal restoration. *Ecological Economics,* 100, pp. 74-84.

CALLICOTT, J.B., 1984. Non-anthropocentric value theory and environmental ethics. *American Philosophical Quarterly,* 21(4), pp. 299-309.

CARLSSON, F. and JOHANSSON-STENMAN, O., 2012. Behavioral economics and environmental policy. *Annual Review of Resource Economics,* 4(1), pp. 75-99.

CARPENTER, S.R., DEFRIES, R., DIETZ, T., et al., 2006. Millennium Ecosystem Assessment: Research needs. *Science,* 314(5797), pp. 257.

CENTEMERI, L., 2009. Environmental damage as negative externality: uncertainty, moral complexity and the limits of the market. *e-cadernos CES,* 05 (2009), pp. 21-40.

CHAIKUMBUNG, M., DOUCOULIAGOS, H. and SCARBOROUGH, H., 2016. The economic value of wetlands in developing countries: A meta-regression analysis. *Ecological Economics,* 124, pp. 164-174.

CHAMP, P.A., BOYLE, K.J. and BROWN, T.C., eds, 2003. *A Primer on Nonmarket Valuation.* Dortdrecht: Springer.

CHAN, K.M., GOULD, R.K. and PASCUAL, U., 2018. Editorial overview: Relational values: what are they, and what's the fuss about? *Current Opinion in Environmental Sustainability,* 36, pp. A1–A7.

CHAN, K.M.A. and SATTERFIELD, T., 2020. The maturation of ecosystem services: Social and policy research expands, but whither biophysically informed valuation? *People and Nature,* 2(4), pp. 1021-1060.

CHAN, K.M.A., BALVANERA, P., BENESSAIAH, K., et al., 2016. Opinion: Why protect nature? Rethinking values and the environment. *Proceedings of the National Academy of Sciences,* 113(6), pp. 1462-1465.

CHAN, K.M.A., GUERRY, A.D., BALVANERA, P., et al., 2012b. Where are cultural and social in ecosystem services? A Framework for constructive engagement. *Bioscience,* 62(8), pp. 744-756.

CHAN, K.M.A., SATTERFIELD, T. and GOLDSTEIN, J., 2012a. Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics,* 74, pp. 8-18.

CHEE, Y.E., 2004. An ecological perspective on the valuation of ecosystem services. *Biological Conservation,* 120(4), pp. 549-565.

CHEGE, J., 2009. *Valuing Preferences for Freshwater Inflows into Five Eastern Cape and Kwazulu-Natal Estuaries*. Gqeberha, South Africa: Faculty of Economic Sciences, Nelson Mandela Metropolitan University.

CHEN, Z.M., CHEN, G.Q., CHEN, B., ZHOU, J.B., YANG, Z.F. and ZHOU, Y., 2009. Net ecosystem services value of wetland: Environmental economic account. *Communications in Nonlinear Science and Numerical Simulation,* 14(6), pp. 2837-2843.

CHRISTIE, M., FAZEY, I., COOPER, R., HYDE, T. and KENTER, J.O., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. *Ecological Economics,* 83, pp. 67-78.

CILLIERS, P., 2000. What can we learn from a theory of complexity? *Emergence,* 2(1), pp. 23-33.

CLARKSON, B., 2012. Wetland restoration monitoring. In: M. PETERS and B. CLARKSON, eds, *Wetland Restoration: A Handbook for New Zealand Freshwater Systems.* Lincoln: Manaaki Whenua Press, pp. 242-261.

CLARO, E., 2007. Exchange relationships and the environment: The acceptability of compensation in the siting of waste disposal facilities. *Environmental Values,* 16(2), pp. 187-208.

CLEWELL, A.F. and ARONSON, J., 2006. Motivations for the restoration of ecosystems. *Conservation Biology*, 20(2), pp. 420-428.

CLEWELL, A.F. and ARONSON, J., 2007. *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession.* 2 edn. Washington, D.C.: Island Press.

CLEWELL, A.F., RIEGER, J. and MUNRO, J., 2005. *Society for Ecological Restoration International Guidelines for Developing and Managing Ecological Restoration Projects.* 2 edn. Tucson: Society for Ecological Restoration.

COASE, R.H., 1978. Economics and contiguous disciplines. *The Journal of Legal Studies,* 7(2), pp. 201-211.

CONRADIE, D.C.U., 2012. *South Africa's Climatic Zones: Today, Tomorrow.* International Green Building Conference and Exhibition: Future Trends and Issues Impacting on the Built Environment, 25-26 July 2012. Sandton.

CONVENTION ON BIOLOGICAL DIVERSITY and UNITED NATIONS ENVIRONMENT PROGRAMME (UNEP), 2018. *Pan-African Action Agenda on Ecosystem Restoration for Increased Resilience.* Kenya: United Nations Environment Programme.

COOK, C.N., MASCIA, M.B., SCHWARTZ, M.W., POSSINGHAM, H.P. and FULLER, R.A., 2013. Achieving conservation science that bridges the knowledge–action boundary. *Conservation Biology,* 27(4), pp. 669-678.

COSTANZA, R. and DALY, H.E., 1992. Natural capital and sustainable development. *Conservation Biology,* 6(1), pp. 37-46.

COSTANZA, R. and FOLKE, C., 1997. Valuing ecosystem services with efficiency, fairness and sustainability as goals. In: G. DAILY, ed, *Nature's Services: Societal Dependence on Natural Ecosystems.* Washington, D.C.: Island Press, pp. 49-70.

COSTANZA, R., 1989. What is ecological economics? *Ecological Economics,* 1(1), pp. 1-7.

COSTANZA, R., 2000. Social goals and the valuation of ecosystem services. *Ecosystems,* 3(1), pp. 4-10.

COSTANZA, R., CUMBERLAND, J., DALY, H., GOODLAND, R. and NORGAARD, R., 1997. *An Introduction to Ecological Economics.* Florida: St. Lucie Press and ISEE.

COSTANZA, R., DE GROOT, R., BRAAT, L., KUBISZEWSKI, I., FIORAMONTI, L., SUTTON, P., FARBER, S. and GRASSO, M., 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosystem Services,* 28, pp. 1-16.

COSTANZA, R., PERRINGS, C. and CLEVELAND, C.J., 1997. Historical roots and motivations. In: R. COSTANZA, C. PERRINGS and C.J. CLEVELAND, eds, *The Development of an Ecological Economics.* Cheltenham: Edward Elgar, pp. 45-67.

COWDEN, C. and KOTZE, D.C., 2008. *WETRehabEvaluate: Guidelines for the Monitoring and Evaluation of Wetland Rehabilitation Projects.* WRC Report TT 342/08. Pretoria: Water Research Commission.

COWDEN, C., KOTZE, D. and PIKE, T., 2013. *Assessment of the Long-Term Response of Two Wetlands to Working for Wetlands Rehabilitation.* WRC Report 2035/1/13. Pretoria: Water Research Commission.

CRAFFORD, J.G. and HASSAN, R.M., 2014. Towards measuring relationships between ecological infrastructure and the economy: The case of a fishery. *South African Journal of Science,* 110(7/8), pp. 1-8.

CRÉPIN, A., BIGGS, R., POLASKY, S., TROELL, M. and DE ZEEUW, A., 2012. Regime shifts and management. *Ecological Economics,* 84, pp. 15-22.

CROOKES, D., 2003. The contribution of livelihood activities in the Limpopo province: Case study evidence from Makua and Manganeng. *Development Southern Africa,* 20(1), pp. 143-159.

CROWARDS, T., 1997. Nonuse values and the environment: Economic and ethical motivations. *Environmental Values,* 6(2), pp. 143-167.

DADA, R., KOTZE D., ELLERY W. and UYS M., 2007. *WET-RoadMap: A Guide to the Wetland Management Series.* WRC Report No TT 321/07. Pretoria: Water Research Commission.

DALY, H.E., 1992. Allocation, distribution, and scale: towards an economics that is efficient, just, and sustainable. *Ecological Economics,* 6(3), pp. 185-193.

DAVIES, G.R., 2013. Appraising weak and strong sustainability: searching for a middle ground. *Consilience,* (10), pp. 111-124.

DAVIS, J.B., 2006. The turn in economics: neoclassical dominance to mainstream pluralism? *Journal of Institutional Economics,* 2(01), pp. 1-20.

DAY, J.W., HALL, C.A., YÁÑEZ-ARANCIBIA, A., PIMENTEL, D., MARTÍ, C.I. and MITSCH, W.J., 2009. Ecology in times of scarcity. *Bioscience,* 59(4), pp. 321-331.

DE GROOT, R., BRANDER, L., VAN DER PLOEG, S., et al., 2012. Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services,* 1(1), pp. 50-61.

DE GROOT, R., FISHER, B., CHRISTIE, M., et al., 2010. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: P. KUMAR, ed, *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations.* London: Earthscan.

DE GROOT, R.S., BLIGNAUT, J., VAN DER PLOEG, S., et al., 2013. Benefits of investing in ecosystem restoration. *Conservation Biology,* 27(6), pp. 1286-1293.

DE GROOT, R.S., STUIP, M.A.M., FINLAYSON, C.M. and DAVIDSON, N., 2006. *Valuing Wetlands: Guidance for Valuing the Benefits Derived from Wetland Ecosystem Services.* Ramsar Technical Report No.3/CBD Technical Series No. 27. Gland & Montreal: Ramsar Convention Secretariat & Secretariat of the Convention on Biological Diversity.

DE GROOT, R.S., WILSON, M.A. and BOUMANS, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics,* 41(3), pp. 393-408.

DE LANGE, W.J., MAHUMANI, B.K., STEYN, M. and OELOFSE, S.H.H., 2012. Monetary valuation of salinity impacts and microbial pollution in the Olifants Water Management Area, South Africa. *Water SA,* 38(2), pp. 241-248.

DE LEO, G.A. and LEVIN, S., 1997. The multifaceted aspects of ecosystem integrity. *Conservation Ecology,* 1(1), pp. 22.

DE WIT, M.P., 2019. The shifting mind of economics. In: J.V. VISSER and M. VISSER, eds, *Seeking Understanding the Lifelong Pursuit to Build the Scientific Mind.* Leiden: Brill, pp. 246-261.

DELANG, C.O., 2006. Not just minor forest products: The economic rationale for the consumption of wild food plants by subsistence farmers. *Ecological Economics,* 59(1), pp. 64-73.

DENIS, P., 2013. The churches' response to political violence in the last years of apartheid: the case of Mpophomeni in the Natal Midlands. *Studia Historiae Ecclesiasticae,* 39(1), [online].

DENYER, K.Y., AKOIJAM, Y., ALI, M., et al., 2018. *Learning from Experience: How Indigenous Peoples and Local Communities Contribute to Wetland Conservation in Asia and Oceania.* Gland: Ramsar Convention Secretariat.

DEPARTMENT FOR ENVIRONMENT, FOOD AND RURAL AFFAIRS (DEFRA), 2011. *An Introductory Guide to Valuing Ecosystem Services.* Report No. PB12852. London: Department for Environment, Food and Rural Affairs, United Kingdom.

DEPARTMENT OF ENVIRONMENT, FORESTRY AND FISHERIES (DEFF), 2021. *Working for Wetlands: 20 Years of Wetland Restoration in South Africa.* Pretoria: Department of Environment, Forestry and Fisheries (DEFF), Republic of South Africa.

DEPARTMENT OF ENVIRONMENTAL AFFAIRS (DEA), 2010. *Green Economy Summit Report.* Pretoria: Department of Environmental Affairs (now Department of Environment, Forestry and Fisheries), Republic of South Africa.

DEPARTMENT OF ENVIRONMENTAL AFFAIRS (DEA), 2018. *South Africa: Final Country Report of the Land Degradation Neutrality Target Setting Programme.* Pretoria: Department of Environmental Affairs (now Department of Environment, Forestry and Fisheries), Republic of South Africa.

DEPARTMENT OF WATER AFFAIRS AND FORESTRY (DWAF), 1996. *South African Water Quality Guidelines: Volume 7: Aquatic Ecosystems.* 1 edn. Pretoria: Department of Water Affairs and Forestry (now Department of Water and Sanitation), Republic of South Africa.

DEPARTMENT OF WATER AFFAIRS AND FORESTRY (DWAF), 2002. *National Eutrophication Monitoring Programme Implementation Manual.* 1 edn. Pretoria: Department of Water Affairs and Forestry (now Department of Water and Sanitation), Republic of South Africa.

DEPARTMENT OF WATER AND SANITATION (DWS), 2018. *National Water and Sanitation Master Plan: Volume 1 Call to Action.* Pretoria: Department of Water and Sanitation, Republic of South Africa.

DEPIETRI, Y. and MCPHEARSON, T., 2017. Integrating the grey, green, and blue in cities: Nature-based solutions for climate change adaptation and risk Reduction. In: N. KABISCH, H. KORN, J. STADLER and A. BONN, eds, *Nature-Based Solutions to Climate Change Adaptation in Urban Areas: Linkages between Science, Policy and Practice.* Cham: Springer International Publishing, pp. 91-109.

DÍAZ, S., DEMISSEW, S., CARABIAS, J., et al., 2015. The IPBES Conceptual Framework — connecting nature and people. *Current Opinion in Environmental Sustainability,* 14, pp. 1-16.

DÍAZ, S., PATAKI, G., ROTH, E., et al., 2014. *Preliminary Guide Regarding Diverse Conceptualization of Multiple Values of Nature and its Benefits, Including Biodiversity and Ecosystem Functions and Services.* Report No. IPBES/4/INF/13. [Online] Available at: https://ipbes.net/diverse-values-valuation: Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

DICKIE, M., Defensive behavior and damage cost methods. In: P.A. CHAMP, K.J. BOYLE and T.C. BROWN, eds, *A Primer on Nonmarket Valuation.* Dortdrecht: Springer, pp. 395-444.

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

DIETZ, T., FITZGERALD, A. and SHWOM, R., 2005. Environmental values. *Annual Review of Environment & Resources,* 30(1), pp. 335-372.

DIXON, J. and PAGIOLA, S., 1998. *Economic Analysis and Environmental Assessment.* Environmental assessment sourcebook update. Washinton, D.C.: The World Bank.

DOAK, D.F., BAKKER, V.J., GOLDSTEIN, B.E. and HALE, B., 2014. What is the future of conservation? *Trends in Ecology & Evolution*, 29(2): 77-81.

DOPFER, K., 1988. Classical mechanics with an ethical dimension: Professor Tinbergen's economics. *Journal of Economic Issues,* 22(3), pp. 675-706.

DUNFORD, R., HARRISON, P., SMITH, A., et al., 2018. Integrating methods for ecosystem service assessment: Experiences from real world situations. Ecosystem Services, pp. 11-21.

DUZI UMNGENI CONSERVATION TRUST (DUCT), 2018. *Save Midmar Project Close-out Report.* Close out Report for Period October 2015 to September 2018 submitted to uMgungundlovu District Municipality. Pietermaritzburg: Duzi uMngeni Conservation Trust.

EARNHART, D., 2001. Combining revealed and stated preference methods to value environmental amenities at residential locations. *Land Economics,* 77(1), pp. 12-29.

EFTEC, 2006. *Valuing Our Natural Environment. Report to Department for Environment, Food and Rural Affairs.* Report No. NR0103. London: eftec.

EHRLICH, P.R. and PRINGLE, R.M., 2008. Where does biodiversity go from here? A grim business-as-usual forecast and a hopeful portfolio of partial solutions. *Proceedings of the National Academy of Sciences,* 105(Supplement 1), pp. 11579-11586.

EKINS, P., DRESNER, S. and DAHLSTRÖM, K., 2008. The four-capital method of sustainable development evaluation. *European Environment,* 18(2), pp. 63-80.

EKINS, P., SIMON, S., DEUTSCH, L., FOLKE, C. and DE GROOT, R., 2003. A framework for the practical application of the concepts of critical natural capital and strong sustainability. *Ecological Economics,* 44(2): 165-185.

EKSTEDT, H. and FUSARI, A., 2010. *Economic Theory and Social Change, Problems and Revisions.* London: Routledge.

ELLIOT, R., 1982. Faking nature. *Inquiry: An Interdisciplinary Journal of Philosophy,* 25(1), pp. 81-93.

EMERTON, L. and KEKULANDALA, L. D. C. B., 2003. *Assessment of the Economic Value of Muthurajawela Wetland.* Occasional Paper IUCN Sri Lanka, 4. Colombo: International Union for Conservation of Nature (IUCN).

EMERTON, L., IYANGO, L., LUWUM, P. and MALINGA, A., 1998. *The Present Economic Value of Nakivubo Urban Wetland, Uganda.* Nairobi: International Union for Conservation of Nature (IUCN).

EPA (UNITED STATES ENVIRONMENTAL PROTECTION AGENCY), 2009. *Valuing the Protection of Ecological Systems and Services. A Report of the EPA Science and Advisory Board.* Report No. EPA-SAB-09-012. Washington, D.C.: United States Environmental Protection Agency.

EPPINK, F.V., BRANDER, L.M. and WAGTENDONK, A.J., 2014. An initial assessment of the economic value of coastal and freshwater wetlands in west Asia. *Land,* 3(3), pp. 557-573.

ESLER, K.J., DOWNSBOROUGH, L., ROUX, D.J., et al., 2016. Interdisciplinary and multi-institutional higher learning: Reflecting on a South African case study investigating complex and dynamic environmental challenges. *Current Opinion in Environmental Sustainability,* 19, pp. 76-86.

EVERARD, M., 2015. Community-based groundwater and ecosystem restoration in semi-arid north Rajasthan: Socio-economic progress and lessons for groundwater-dependent areas. *Ecosystem Services,* 16, pp. 125-135.

EVERARD, M., KATARIA, G., KUMAR, S. and GUPTA, N., 2021. Assessing livelihood-ecosystem interdependencies and natural resource governance in a tribally controlled region of India's north-eastern middle Himalayas. *Environment, Development and Sustainability,* 23(5), pp. 7772-7790.

FANNY, B., NICOLAS, D., SANDER, J., ERIK, G. and MARC, D., 2015. How (not) to perform ecosystem service valuations: pricing gorillas in the mist. *Biodiversity and Conservation,* 24(1), pp. 187-197.

FAO, IUCN, CEM and SER, 2021. *Principles for Ecosystem Restoration to Guide the United Nations Decade 2021–2030.* Rome: Food and Agricultural Organization of the United Nations (FAO).

FARBER, S.C., COSTANZA, R. and WILSON, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. *Ecological Economics,* 41(3), pp. 375-392.

FARLEY, J., 2008. The role of prices in conserving critical natural capital. *Conservation Biology,* 22(6), pp. 1399-1408.

FARLEY, J., 2012. Ecosystem services: The economics debate. *Ecosystem Services,* 1(1), pp. 40-49.

FELTON, I., 2017. *Mthinzima Wetland Rehabilitation*. Environmental Planning. Pietermaritzburg: KZN Department of Economic Development, Tourism & Environmental Affairs. Personal Communication, 29 January.

FENICHEL, E.P. and ABBOTT, J.K., 2014. Natural capital: from metaphor to measurement. *Journal of the Association of Environmental and Resource Economists,* 1(1), pp. 1-27.

FENICHEL, E.P., KOTCHEN, M.J. and ADDICOTT, E.T., 2017. *Even the Representative Agent Must Die: Using Demographics to Inform Long-Term Social Discount Rates.* Cambridge, Massachusetts: National Bureau of Economic Research, Inc.

FIGUEROA, E.B., 2007. Restoring natural capital: A mainstream economic perspective. In: A. ARONSON, S.J. MILTON and J.N. BLIGNAUT, eds, *Restoring Natural Capital, Science, Business and Practice.* Washington D.C.: Island Press, pp. 28-35.

FISCHENICH, C., 2008. *The Application of Conceptual Models to Ecosystem Restoration.* Report No. ERDC TN-EMRRP-EBA-01. Vicksburg, Mississippi: ERDC Environmental Laboratory.

FISCHER, F., 1998. Policy inquiry in post-positivist perspective. *Policy Studies Journal,* 26(1), pp. 129-146.

FISHER, B., TURNER, K., ZYLSTRA, M., et al., 2008. Ecosystem services and economic theory: Integration for policy-relevant research. *Ecological Applications,* 18(8), pp. 2050-2067.

FISHER, B., TURNER, R.K. and MORLING, P., 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics,* 68(3), pp. 643-653.

FOSTER, J.B. and CLARK, B., 2009. The Paradox of wealth: Capitalism and ecological destruction. *Monthly Review: An Independent Socialist Magazine,* 61(6), pp. 1-18.

FOURIE, H., DE WIT, M. and VAN DER MERWE, A., 2013. The role and value of water in natural capital restoration on the Aagulhas Plain. *South African Journal of Economic and Management Sciences,* 16(1), pp. 83-95.

FREDERICK, S., 2006. Valuing future life and future lives: A framework for understanding discounting. *Journal of Economic Psychology,* 27(5), pp. 667-680.

FREEMAN, M., 2003. Economic valuation: What and why. In: P.A. CHAMP, K.J. BOYLE and T.C. BROWN, eds, *A Primer on Nonmarket Valuation.* Dortdrecht: Springer, pp. 1-26.

FREY, B.S. and JEGEN, R., 2001. Motivation Crowding Theory. *Journal of Economic Surveys,* 15(5), pp. 589-611.

FULLBROOK, E., 2009. Introduction: Lawson's reorientation. In: E. FULLBROOK, ed, *Ontology and Economics.* Abingdon, London: Routledge, pp. 1-12.

GANN, G.D., MCDONALD, T., WALDER, B., et al., 2019. International principles and standards for the practice of ecological restoration. Second edition. *Restoration Ecology,* 27, pp. S1-S46.
GEORGESCU-ROEGEN, N., 1975. Energy and economic myths. *Southern Economic Journal,* 41(3), pp. 347-381.
GHERMANDI, A., VAN DEN BERGH, J.C.J.M., BRANDER, L.M., et al., 2010. Values of natural and human-made wetlands: A meta-analysis. *Water Resources Research,* 46(W12516), pp. 1-12.
GINSBURG, A.E., CRAFFORD, J.G. and HARRIS, K.R., 2010. *Framework and Manual for the Evaluation of Aquatic Ecosystems Services for the Resource Directed Measures.* WRC Report No. TT 462/10. Pretoria: Water Research Commission.
GINTIS, H., 2000. Beyond Homo economicus: Evidence from experimental economics. *Ecological Economics,* 35(3), pp. 311-322.
GOLDSTEIN, J.H., PEJCHAR, L. and DAILY, G.C., 2008. Using return-on-investment to guide restoration: a case study from Hawaii. *Conservation Letters,* 1, pp. 236-243.
GOLLIER, C., 2010. Ecological discounting. *Journal of Economic Theory,* 45(2), pp. 812-829.
GÓMEZ-BAGGETHUN, E. and BARTON, D.N., 2013. Classifying and valuing ecosystem services for urban planning. *Ecological Economics,* 86, pp. 235-245.
GÓMEZ-BAGGETHUN, E. and DE GROOT, R.D., 2010. Natural capital and ecosystem services: The ecological foundation of human society. In: R.M. HARRISON and R.E. HESTER, eds, *Ecosystem Services*. London: The Royal Society of Chemistry, pp. 105-121.
GÓMEZ-BAGGETHUN, E. and RUIZ-PÉREZ, M., 2011. Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography,* 35(5), pp. 613-628.
GÓMEZ-BAGGETHUN, E., DE GROOT, R., LOMAS, P.L. and MONTES, C., 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics,* 69(6), pp. 1209-1218.
GÓMEZ-BAGGETHUN, E., MARTÍN-LÓPEZ, B., BARTON, D., et al., 2014. *State-Of-The-Art Report on Integrated Valuation of Ecosystem Services. EU FP7 OpenNESS Project Deliverable 4.1.* [Online] Available at http://www.openness-project.eu/: European Commission.
GOULDER, L.H. and KENNEDY, D., 1997. *Valuing Ecosystem Services: Philosophical Bases and Empirical Methods.* Washington, D.C.: Island Press.
GOWDY, J., 2004. Altruism, evolution, and welfare economics. *Journal of Economic Behavior & Organization,* 53, pp. 69-73.
GOWDY, J., HOWARTH, R.B. and TISDELL, C., 2010. Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity. In: P. KUMAR, ed, *The Economics of Ecosystems and Biodiversity (TEEB) Ecological and Economic Foundations.* London: Earthscan, pp. Chapter 6.
GRAHAM, M., BLIGNAUT, J. and DE VILLIERS, LOUIS ET AL., 2012. *Development of a Generic Model to Assess the Costs Associated with Eutrophication.* WRC Report No. 1568/1/12. Pretoria: Water Research Commission.
GRAHAM, P.M., 2004. *Modelling the Water Quality in Dams within the Umgeni Water Operational Area with Emphasis on Algal Relations.* PhD Thesis. Potchefstroom: Department of Botany, North-West University.
GREN, I., 1995. Costs and benefits of restoring wetlands: Two Swedish case studies. *Ecological Engineering,* 4(2), pp. 153-162.
GRENFELL, M.C., ELLERY, W.N., GARDEN, S.E., DINI, J. and VAN DER VALK, A. G., 2007. The language of intervention: A review of concepts and terminology in wetland ecosystem repair. *Water SA,* 33(1), pp. 43-50.
GREY, W., 1993. Anthropocentrism and deep ecology. *Australasian Journal of Philosophy,* 71(4), pp. 463-475.
GROOM, B. and TURK, Z., 2021. Reflections on the Dasgupta Review on the Economics of Biodiversity. *Environmental and Resource Economics,* 79(1), pp. 1-23.
GROSSMAN, M., 2012. *Economic Valuation of Wetland Ecosystem Services. Accounting for Wetland Ecosystem Service Benefits in Cost Benefit Analysis of River Basin Management Options with Case*

Studies from the Elbe River Basin. PhD Thesis. Berlin: Faculty of Planning, Construction and Environment, Technical University of Berlin.
GROUNDTRUTH, 2010. *Wetland Rehabilitation Plan for Edendale Shopping Mall.* Report No. GTW053-100610-01. Hilton, South Africa: GroundTruth Water, Wetlands, Biodiversity and Environmental Engineering.
GROUNDTRUTH, 2012. *Upper uMngeni Integrated Catchment Management Plan.* Report No. GT0165-0812. Hilton, South Africal: GroundTruth Water, Wetlands, Biodiversity and Environmental Engineering.
GROUNDTRUTH, 2015. *Wetland Rehabilitation Plan: uMthinzima Stream Wetland Downstream of R617 Road Howick.* Report No. GTW257-010915-01. Hilton, South Africa: GroundTruth Water, Wetlands, Biodiversity and Environmental Engineering.
GULL, K., 2012. *An Economic Case Study of Land Rehabilitation in the Kromme River Catchment*. MSc Thesis. Cape Town: Economics Department, University of Cape Town.
GUNDERSON, L. and LIGHT, S.S., 2006. Adaptive management and adaptive governance in the Everglades ecosystem. *Policy Sciences,* 39(4), pp. 323-334.
GUNDERSON, L.H., CARPENTER, S.R., FOLKE, C., OLSSON, P. and PETERSON, G.D., 2006. Water RATs (resilience, adaptability, and transformability) in lake and wetland and social-ecological systems. *Ecology and Society,* 11(1), pp. 16-26.
GUTRICH, J.J. and HITZHUSEN, F.J., 2004. Assessing the substitutability of mitigation wetlands for natural sites: estimating restoration lag costs of wetland mitigation. *Ecological Economics,* 48(4), pp. 409-424.
HAINES-YOUNG, R. and POTSCHIN, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: D. RAFFAELLI and C. FRID, eds, *Ecosystem Ecology: A New Synthesis, BES Ecological Reviews Series.* Cambridge: Cambridge University Press, pp. 110-139.
HAINES-YOUNG, R. and POTSCHIN, M., 2013. *Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012.* Nottingham: Centre for Environmental Management, University of Nottingham.
HANEMANN, W.M., 1994. Valuing the environment through contingent valuation. *The Journal of Economic Perspectives,* 8(4), pp. 19-43.
HANLEY, N. and BARBIER, E.B., 2009. *Pricing Nature: Cost-benefit Analysis and Environmental Policy.* Cheltenham: Edward Elgar.
HANSJÜRGENS, B., SCHRÖTER-SCHLAACK, C., BERGHÖFER, A. and LIENHOOP, N., 2017. Justifying social values of nature: Economic reasoning beyond self-interested preferences. *Ecosystem Services,* 23, pp. 9-17.
HARRISON, P.A., DUNFORD, R., BARTON, D.N., et al., 2018. Selecting methods for ecosystem service assessment: A decision tree approach. *Ecosystem Services,* 29, pp. 481-498.
HAUSMAN, D. and MCPHERSON, M., 2008. The philosophical foundations of mainstream normative economics. In: D. HAUSMAN, ed, *The Philosophy of Economics An Anthology.* 3 edn. Cambridge: Cambridge University Press, pp. 226-250.
HAY, D., KOTSE, D. and BREEN, C., 2014. *WELL-WET Wetlands and Wellbeing: Getting More out of South Africa's Wetlands. An Introductory Handbook.* WRC Report TT 605/14. Pretoria: Water Research Commission.
HAYWARD, T., 1997. Anthropocentrism: A misunderstood problem. *Environmental Values,* 6(1), pp. 49-63.
HE, J., AI, J., ZHU, X. and SUN, X., 2015. Ecological compensation standards of wetland restoration projects. *Polish Journal of Environmental Studies,* 24(6), pp. 2421-2432.
HEAL, G.M., BARBIER, E.B., BOYLE, K.J., et al., 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making.* Committee on Assessing and Valuing the Services of Aquatic and Related Terrestrial Ecosystems. Washington, D.C.: National Academies Press.
HEIN, L., VAN KOPPEN, K., DE GROOT, R.S. and VAN IERLAND, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics,* 57(2), pp. 209-228.

HEINZEN, B., 2004. *Feeling for Stones: Learning and Invention When Facing the Unknown.* London: Barbara Heinzen.

HENRICH, J., BOYD, R., BOWLES, S., CAMERER, C., FEHR, E., GINTIS, H. and MCELREATH, R., 2001. In search of Homo Economicus: Behavioral experiments in 15 small-scale societies. *American Economic Review,* 91(2), pp. 73-78.

HERMANS, L.M., NABER, A.C. and ENSERINK, B., 2012. An approach to design long-term monitoring and evaluation frameworks in multi-actor systems—A case in water management. *Evaluation and Program Planning*, 35(4), pp. 427-438.

HETTINGER, N., 2012. Nature Restoration as a paradigm for the human relationship with nature. In: A. THOMPSON and J. BENDIK-KEYMER, eds, *Ethical Adaptation to Climate Change.* Cambridge, Massachusetts: The MIT Press, pp. 27-46.

HIGGS, E., FALK, D.A., GUERRINI, A., et al., 2014. The changing role of history in restoration ecology. *Frontiers in Ecology and the Environment,* 12(9), pp. 499-506.

HIGGS, E.S., 1997. What is good ecological restoration? *Conservation Biology,* 11(2), pp. 338-348.

HODGSON, G.M., 1993. Economics, environmental policy and the transcendence of utilitarianism. In: FOSTER, ed, *Valuing Nature? Ethics, Economics and the Environment.* London: Routledge, pp. 48-66.

HODGSON, G.M., 2003. How Veblenian evolutionary thinking transcends methodological individualism and methodological collectivism. *Économie et institutions,* 3, pp. 5-28.

HOFFMANN-RIEM, H., BIBER-KLEMM, S., GROSSENBACHER-MANSUY, W., et al., 2008. Idea of the handbook. In: G.H. HADORN, H. HOFFMANN-RIEM, S. BIBER-KLEMM, et al., eds, *Handbook of Transdisciplinary Research.* Berlin: Springer Science and Business Media B.V., pp. 3-18.

HOLLAND, A., 2002. Are choices tradeoffs? In: D.W. BROMLEY and J. PAAVOLA, eds, *Economics, Ethics, and Environmental Policy: Contested Choices.* Oxford: Blackwell Publishers Ltd, pp. 17-34.

HOLLING, C.S., 1996. Surprise for Science, Resilience for Ecosystems, and Incentives for People. *Ecological Applications,* 6(3), pp. 733-735.

HOOVER, K.D., 2008. Does macroeconomics need microfoundations? In: D.M. HAUSMAN, ed, *The Philosophy of Economics, An Anthology.* Third edn. Cambridge: Cambridge University Press, pp. 315-333.

HOSKING, S., 2011. The recreational value of river inflows into South African estuaries. *Water SA,* 37(5), pp. 711-718.

HOUDET, J., TROMMETTER, M. and WEBER, J., 2012. Understanding changes in business strategies regarding biodiversity and ecosystem services. Ecological Economics, 73, pp. 37-16.

HOWARTH, R.B., 1995. Sustainability under uncertainty: A deontological approach. *Land Economics,* 71(4), pp. 417-427.

HUBACEK, K. and VAN DEN BERGH, J.C.J.M., 2006. Changing concepts of 'land' in economic theory: From single to multi-disciplinary approaches. *Ecological Economics,* 56(1), pp. 5-27.

IFTEKHAR, M.S., POLYAKOV, M., ANSELL, D., GIBSON, F. and KAY, G.M., 2017. How economics can further the success of ecological restoration. *Conservation Biology,* 31(2), pp. 261-268.

INTERGOVERNMENTAL SCIENCE-POLICY PLATFORM ON BIODIVERSITY AND ECOSYSTEM SERVICES (IPBES), 2018. *The IPBES Assessment Report on Land Degradation and Restoration.* Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

JACOBS, S., DENDONCKER, N., MARTÍN-LÓPEZ, B., et al., 2016. A new valuation school: Integrating diverse values of nature in resource and land use decisions. *Ecosystem Services*, 22, pp. 213-220.

JAX, K., 2014. Thresholds, tipping points and limits. OpenNESS Synthesis Paper No 4. In: M. POTSCHIN and K. JAX, eds, *OpenNESS Reference Book. EC FP7 Grant Agreement no. 308428.* Helsinki: Finnish Environment Institute, pp. 1-4.

JAX, K., BARTON, D.N., CHAN, K.M.A., et al., 2013. Ecosystem services and ethics. *Ecological Economics,* 93, pp. 260-268.

JAX, K., CALESTANI, M., CHAN, K.M., et al., 2018. Caring for nature matters: a relational approach for understanding nature's contributions to human well-being. *Current Opinion in Environmental Sustainability*, 35, pp. 22-29.

JENKINS, W.A., MURRAY, B.C., KRAMER, R.A. and FAULKNER, S.P., 2010. Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. *Ecological Economics,* 69, pp. 1051-1061.

JOGIAT, R., 2013. *Save the Midmar Dam Project.* Project brief. Pietermaritzburg: uMgungundlovu District Municipality.

KADLEC, R.H. and WALLACE, S.D., 2009. *Treatment Wetlands.* 2 edn. Florida: CRC Press.

KAHNEMAN, D. and TVERSKY, A., 1979. Prospect Theory: An analysis of decision under risk. *Econometrica,* 47(2), pp. 263-291.

KAHNEMAN, D., 2003. Maps of bounded rationality: Psychology for behavioral economics. *American Economic Review,* 93(5), pp. 1449-1475.

KALLIS, G., GÓMEZ-BAGGETHUN, E. and ZOGRAFOS, C., 2013. To value or not to value? That is not the question. *Ecological Economics,* 94, pp. 97-105.

KANGALAWE, R.Y.M. and LIWENGA, E.T., 2005. Livelihoods in the wetlands of Kilombero Valley in Tanzania: Opportunities and challenges to integrated water resource management. *Physics and Chemistry of the Earth,* 30(11), pp. 968-975.

KATZ, E., 1992. The big lie: Human restoration of nature. In: D.M. KAPLAN, ed, *Readings in the Philosophy of Technology.* 2 edn. Lanham, Maryland: Rowman & Littlefield, pp. 390-397.

KATZ, E., 1999. A pragmatic reconsideration of anthropocentrism. *Environmental Ethics,* 21(4), pp. 377-390.

KATZ, E., 2011. Envisioning a de-anthropocentrised world: Critical comments on Anthony Weston's 'The Incompleat Eco-Philosopher'. *Ethics, Policy & Environment,* 14(1), pp. 97-101.

KATZ, E., 2012. Further adventures in the case against restoration. *Environmental Ethics,* 34(1), pp. 67-97.

KAY, J.J., 1991. A nonequilibrium thermodynamic framework for discussing ecosystem integrity. *Environmental management,* 15(4), pp. 483-495.

KEELER, B.L., POLASKY, S., BRAUMAN, K.A., et al., 2012. Linking water quality and well-being for improved assessment and valuation of ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America,* 109(45), pp. 18619-18624.

KENTER, J.O., BRYCE, R., CHRISTIE, M., et al., 2016. Shared values and deliberative valuation: Future directions. *Ecosystem Services,* 21, pp. 358-371.

KENTER, J.O., HYDE, T., CHRISTIE, M. and FAZEY, I., 2011. The importance of deliberation in valuing ecosystem services in developing countries—Evidence from the Solomon Islands. *Global Environmental Change,* 21(2), pp. 505-521.

KENTER, J.O., O'BRIEN, L., HOCKLEY, N., et al., 2015. What are shared and social values of ecosystems? *Ecological Economics,* 111, pp. 86-99.

KHOSRAVI, F. and JHA-THAKUR, U., 2019. Managing uncertainties through scenario analysis in strategic environmental assessment. *Journal of Environmental Planning and Management,* 62(6), pp. 979-1000.

KLAMER, A., 2002. Accounting for social and cultural values. *De Economist,* 150(4), pp. 453-473.

KLAMER, A., 2003. A pragmatic view on values in economics. *Journal of Economic Methodology,* 10(2), pp. 191-212.

KLEYNHANS, M., TURPIE, J., RUSINGA, F. and GÖRGENS, A., 2010. *Quantification of the Flow Regulation Services Provided by Nylsvley Wetland, South Africa.* WRC Report no. TT 441/09. Pretoria: Water Research Commission.

KOLBE, A.C., 2014. *Citizen Science and Water Quality in the Umngeni Catchment Area, Kwazulu-Natal, South Africa.* MSc Thesis. Ontario: School of Environmental Studies, Queen's University.

KOTZE, D., MEMELA, B., FUZANI, N. and THOBELA, N., 2002. *Utilisation of the Mbongolwane Wetland in KwaZulu-Natal, South Africa.* Report to the International Water Management Institute. Pretoria: International Water Management Institute.

KOTZE, D.C. and ELLERY, W.N., 2008. *WET-OutcomeEvaluate: An Evaluation of the Rehabilitation Outcomes at Six Wetland Sites in South Africa.* WRC Report TT 343/08. Pretoria: Water Research Commission.

KOTZE, D.C., ELLERY, W.N., ROUNTREE, M., et al., 2009. *WET-RehabPlan: Guidelines for Planning Wetland Rehabilitation in South Africa.* WRC Report TT 336/09. Pretoria: Water Research Commission.

KOTZE, D.C., MACFARLANE, D.M., EDWARDS, R.J. and MADIKIZELA, B., 2020. WET-EcoServices Version 2: A revised ecosystem services assessment technique, and its application to selected wetland and riparian areas. *Water SA,* 46(4), pp. 679–688.

KOTZE, D.C., MARNEWECK, G.C., BATCHELOR, A.L., LINDLEY, D.S. and COLLINS, N.B., 2008. *WET-EcoServices: A Technique for Rapidly Assessing Ecosystem Services Supplied by Wetlands.* WRC Report TT 339/08. Pretoria: Water Research Commission.

KRUTILLA, J.V., 1967. Conservation reconsidered. *The American Economic Review,* 57(4), pp. 777-786.

KUMAR, P., BRONDÍZIO, E., ELMQVIST, T., et al., 2010. Key messages and linkages with national and local policies. In: P. KUMAR, ed, *The Economics of Ecosystems and Biodiversity (TEEB): The Ecological and Economic Foundations.* 1 edn. London: Routledge, pp. 390-422.

KUMAR, R., HORWITZ, P., MILTON, G.R., SELLAMUTTU, S.S., BUCKTON, S.T., DAVIDSON, N.C., PATTNAIK, A.K., ZAVAGLI, M. and BAKER, C., 2011. Assessing wetland ecosystem services and poverty interlinkages: a general framework and case study. *Hydrological Sciences Journal,* 56(8), pp. 1602-1621.

KZN DEPARTMENT OF AGRICULTURE AND ENVIRONMENTAL AFFAIRS (DAEA), 2014. *Environmental Authorisation: Mpophomeni Waste Water Treatment Works and Associated Treated Effluent Pipeline and Wetland Enhancement.* Pietermaritzburg: KZN Department of Agriculture and Environmental Affairs, Republic of South Africa.

LAND, M., GRANÉLI, W., GRIMVALL, A., et al., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence,* 5(1), pp. 9.

LANNAS, K.S.M. and TURPIE, J.K., 2009. Valuing the provisioning services of wetlands: contrasting a rural wetland in Lesotho with a peri-urban wetland in South Africa. *Ecology and Society,* 14(2), pp. 18 [online].

LANTZ, V., BOXALL, P., KENNEDY, M. and WILSON, J., 2013. The valuation of wetland conservation in an urban/peri urban watershed. *Regional Environmental Change,* 13(5), pp. 939-953.

LARSON, A.J., BELOTE, R.T., WILLIAMSON, M.A. and APLET, G.H., 2013. Making monitoring count: Project design for active adaptive management. *Journal of Forestry,* 111(5), pp. 348-356.

LAURANS, Y., RANKOVIC, A., BILLÉ, R., PIRARD, R. and MERMET, L., 2013. Use of ecosystem services economic valuation for decision making: Questioning a literature blindspot. *Journal of environmental management,* 119, pp. 208-219.

LEI, L., JIAN, L., YUTAO, W., NVJIE, W. and RENQING, W., 2011. Cost-benefit Analysis and payments for watershed-scale wetland rehabilitation: a case study in Shandong Province, China. *International Journal of Environmental Research,* 5(3), pp. 787-796.

LESHEM, D., 2016. Retrospectives: What did the ancient Greeks mean by Oikonomia? *Journal of Economic Perspectives,* 30(1), pp. 225-38.

LIENING, A., 2013. The breakdown of the traditional mechanistic worldview, the development of complexity sciences and the pretence of knowledge in economics. *Modern Economy,* 4, pp. 305-319.

LIGHT, A., 2002. Contemporary environmental ethics from metaethics to public philosophy. *Metaphilosophy,* 33(4), pp. 426-449.

LINTON, J. and BUDDS, J., 2014. The hydrosocial cycle: Defining and mobilizing a relational-dialectical approach to water. *Geoforum,* 57, pp. 170-180.

LIQUETE, C., UDIAS, A., CONTE, G., GRIZZETTI, B. and MASI, F., 2016. Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits. *Ecosystem Services,* 22, pp. 392-401.

LIU, S., COSTANZA, R., FARBER, S. and TROY, A., 2010. Valuing ecosystem services. *Annals of the New York Academy of Sciences,* 1185(1), pp. 54-78.

LOCKWOOD, M., 1997. Integrated value theory for natural areas. *Ecological Economics,* 20(1), pp. 83-93.

LOCKWOOD, M., 1999. Humans valuing nature: Synthesising insights from philosophy, psychology and economics. *Environmental Values,* 8(3), pp. 381-401.

LUCK, G.W., CHAN, K.M.A., ESER, U., et al., 2012. Ethical considerations in on-ground applications of the ecosystem services concept. *Bioscience,* 62(12), pp. 1020-1029.

MA (MILLENNIUM ECOSYSTEM ASSESSMENT), 2003. *Ecosystems and Human Well-Being: A Framework for Assessment.* Washington, DC: Island Press.

MA (MILLENNIUM ECOSYSTEM ASSESSMENT), 2005a. *Ecosystems and Human Well-Being: Synthesis.* Washington, D.C.: Island Press.

MA (MILLENIUM ECOSYSTEM ASSESSMENT), 2005b. *Ecosystems and Human Well-Being: Wetlands and Water Synthesis.* Washington, D.C.: World Resources Institute.

MACE, G.M., BATEMAN, I., ALBON, S., et al., 2011. Conceptual framework and methodology. In: *The UK*

National Ecosystem Assessment Technical Report. Cambridge: UNEP-WCMC, pp. 11-26.

MACFARLANE, D.M., KOTZE, D.C., ELLERY, W.N., et al., 2008. *WET-Health: A technique For Rapidly Assessing Wetland Health.* WRC Report No. TT 340/09. Pretoria: Water Research Commission.

MAILA, D., MULDERS, J., NAIDOO, N., et al., 2017. *An Evidence-Based Approach to Measuring the Costs and Benefits of Changes in Aquatic Ecosystem Services.* WRC Report No. TT 726/17. Pretoria: Water Research Commission.

MÄKI, U., 2001. Economic ontology: What? Why? How? In: U. MÄKI, ed, *The Economic World View: Studies in the Ontology of Economics.* Cambridge: Cambridge University Press, pp. 3-14.

MÄKI, U., 2011. Scientific realism as a challenge to economics (and vice versa). *Journal of Economic Methodology,* 18(01), pp. 1-12.

MÄKI, U., 2012. Preface. In: U. MÄKI, ed, *Handbook of the Philosophy of Science Volume 13: Philosophy of Economics.* Amsterdam: North Holland, pp. xiii-xvii.

MARSHALL, A., 1890. *Principles of Economics: An Introductory Volume.* 1 edn. London: Macmillan.

MARSHALL, A., 1920. *Principles of Economics: An Introductory Volume.* 8 edn. London: Macmillan and Co.

MARTINEZ-ALIER, J., 2002. *The Environmentalism of the Poor.* Cheltenham: Edward Elgar.

MARTINEZ-ALIER, J., 2008. Languages of valuation. *Economic and Political Weekly,* 43(48), pp. 28-32.

MARTINEZ-ALIER, J., MUNDA, G. and O'NEILL, J., 1998. Weak comparability of values as a foundation for ecological economics. *Ecological Economics,* 26(3), pp. 277-286.

MARTÍN-LÓPEZ, B., GÓMEZ-BAGGETHUN, E., GARCÍA-LLORENTE, M. and MONTES, C., 2014. Trade-offs across value-domains in ecosystem services assessment. *Ecological Indicators,* 37 (Part A), pp. 220-228.

MARTON, J.M., ROY CHOWDHURY, R. and CRAFT, C.B., 2015. A comparison of the spatial variability of denitrification and related soil properties in restored and natural depression wetlands in Indiana, USA. *International Journal of Biodiversity Science, Ecosystem Services & Management,* 11(1), pp. 36-45.

MASIBUMBANE HIV/AIDS MISSION, 2020. *Mpophomeni Background*. Masibumbane HIV/AIDS Mission [Online]. Available: https://www.masibumbane.org.za/background/ [Accessed 1 November, 2020].

MATTHEWS, M.W. and BERNARD, S., 2015. Eutrophication and cyanobacteria in South Africa's standing water bodies: A view from space. *South African Journal of Science,* 111(5), pp. 1-8.

MAX-NEEF, M.A., 2005. Foundations of transdisciplinarity. *Ecological Economics,* 53(1), pp. 5-16.

MCCARTNEY, M., MORARDET, S., REBELO, L., FINLAYSON, C.M. and MASIYANDIMA, M., 2011. A study of wetland hydrology and ecosystem service provision: GaMampa wetland, South Africa. *Hydrological Sciences Journal,* 56(8), pp. 1452-1466.

MCCONNACHIE, M.M., COWLING, R.M., SHACKLETON, C.M. and KNIGHT, A.T., 2013. The challenges of alleviating poverty through ecological restoration: Insights from South Africa's "Working for Water" Program. *Restoration Ecology,* 21(5), pp. 544-550.

MCDONALD, T. and DIXON, K., 2018. National standards: Reasserting the ecological restoration framework in uncertain times. *Ecological Management & Restoration,* 19, pp. 79-89.

MCDONALD, T., GANN, G.D., JONSON, J., DIXON, K.W., 2016. *International standards for the practice of ecological restoration—including principles and key concepts.* Washington, D.C.: Society for Ecological Restoration.

MCGREGOR, S.L.T. and MURNANE, J.A., 2010. Paradigm, methodology and method: intellectual integrity in consumer scholarship. *International Journal of Consumer Studies,* 34(4), pp. 419-427.

MCSHANE, K., 2007. Why Environmental Ethics Shouldn't Give Up on Intrinsic Value. *Environmental Ethics,* 29(1), pp. 43-61.

MELI, P., REY BENAYAS, J.M., BALVANERA, P. and MARTINEZ RAMOS, M., 2014. Restoration enhances wetland biodiversity and ecosystem service supply, but results are context-dependent: A meta-analysis. *Plos One,* 9(4), pp. [online] e93507.

MENARD, C. and SHIRLEY, M.M., 2008. Introduction. In: C. MENARD and M.M. SHIRLEY, eds, *Handbook of New Institutional Economics.* Dortdrecht: Springer, pp. 1-20.

MÉNARD, C., 2011. A new institutional economics perspective on environmental issues. *Environmental Innovation and Societal Transitions,* 1(1), pp. 115-120.

MEYERHOFF, J. and DEHNHARDT, A., 2007. The European Water Framework Directive and economic valuation of wetlands: The restoration of floodplains along the River Elbe. *European Environment,* 17(1), pp. 18-36.

MILLER, T.R., BAIRD, T.D., LITTLEFIELD, C.M., et al., 2008. Epistemological pluralism: Reorganizing interdisciplinary research. *Ecology and Society,* 13(2), pp. 46-63.

MITCHELL, R.C. and CARSON, R.T., 1989. *Using Surveys to Value Public Goods: The Contingent Valuation Method.* 1 edn. New York: Resources for the Future.

MITSCH, W.J. and GOSSELINK, J.G., 2015. *Wetlands.* 5 edn. Hoboken, New Jersey: John Wiley & Sons, Inc.

MORENO-MATEOS, D., POWER, M.E., COMÍN, F.A. and YOCKTENG, R., 2012. Structural and functional loss in restored wetland ecosystems. *PLOS Biology,* 10(1), pp. [online] e1001247.

MORITO, B., 2003. Intrinsic value: A modern albatross for the ecological approach. *Environmental Values,* 12(3), pp. 317-336.

MUIR, J., 1911. *My First Summer in the Sierra.* Sierra Club Books 1988 edition. Boston, New York: Houghton Mifflin.

MULLINS, D., BOTHA, J.P., MOSAKA, D.D., JURGENS, F.X. and MAJORO, T. J. (CONNINGARTH ECONOMISTS), 2014. *A Manual for Cost Benefit Analysis in South Africa with Specific Reference to Water Resource Development. 3rd Edition.* WRC Report TT 598/14. Pretoria: Water Research Commission.

MUNDA, G., 1997. Environmental Economics, Ecological Economics, and the concept of sustainable development. *Environmental Values,* 6(2), pp. 213-233.

MUNDA, G., 2016. Beyond welfare economics: Some methodological issues. *Journal of Economic Methodology,* 23(2), pp. 185-202.

MURACA, B., 2011. The map of moral significance: A new axiological matrix for environmental ethics. *Environmental Values,* 20(3), pp. 375-396.

MURADIAN, R. and PASCUAL, U., 2018. A typology of elementary forms of human-nature relations: a contribution to the valuation debate. *Current Opinion in Environmental Sustainability,* 35, pp. 8-14.

MUSYOKI, A., 2012. *The Emerging Policy for Green Economy and Social Development in Limpopo, South Africa.* Geneva: United Nations Research Institute for Social Development.

MWAKAJE, A.G., 2009. Wetlands, livelihoods and sustainability in Tanzania. *African Journal of Ecology,* 47, pp. 179-184.

NABAHUNGU, N.L. and VISSER, S.M., 2011. Contribution of wetland agriculture to farmers' livelihood in Rwanda. *Ecological Economics,* 71, pp. 4-12.

NAESS, P., 2006a. Cost-Benefit analyses of transportation investments. *Journal of Critical Realism,* 5(1), pp. 32-60.

NAESS, P., 2006b. Unsustainable growth, unsustainable capitalism. *Journal of Critical Realism,* 5(2), pp. 197-227.

NAHLIK, A.M., KENTULA, M.E., FENNESSY, M.S. and LANDERS, D.H., 2012. Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. *Ecological Economics,* 77(0), pp. 27-35.

NAIDOO, N., PEARCE, D., VISSER, W., CRAFFORD, J., MAILA, D. and HARRIS, K., 2016. *Implementation of Effective Wastewater Charges by Municipalities in South Africa: An Investigation into the Barriers and Enablers.* WRC Report No. TT 673/16. Pretoria: Water Research Commission.

NAMUGIZE, J.N., 2017. *Effects of Land Use and Land Cover Changes on Water Quality of the Upper uMngeni River, KwaZulu-Natal Province, South Africa*. PhD Thesis. Pietermaritzburg: School of Agriculture, Earth and Environmental Science, University of KwaZulu-Natal.

NAMUGIZE, J.N., JEWITT, G. and GRAHAM, M., 2018. Effects of land use and land cover changes on water quality in the uMngeni river catchment, South Africa. *Physics and Chemistry of the Earth,* 105, pp. 247-264.

NASSL, M. and LÖFFLER, J., 2015. Ecosystem services in coupled social–ecological systems: Closing the cycle of service provision and societal feedback. *Ambio,* pp. 1-13.

NATIONAL RESEARCH COUNCIL, 1999. *Perspectives on Biodiversity: Valuing Its Role in an Everchanging World.* Washington, D.C.: National Academies Press.

NEL, J.L. and DRIVER, A., 2012. *South African National Biodiversity Assessment 2011: Technical Report. Volume 2: Freshwater Component.* Report No. CSIR/NRE/ECO/IR/2012/0022/A. Stellenbosch: Council for Scientific and Industrial Research.

NELSON, E., MENDOZA, G., REGETZ, J., et al., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment,* 7(1), pp. 4-11.

NELSON, J., 2008. Feminism and economics. In: D.M. HAUSMAN, ed, *The Philosophy Of Economics: An Anthology.* 3 edn. Cambridge: Cambridge University Press, pp. 454-475.

NEUMAYER, E., 2012. Human Development and sustainability. *Journal of Human Development and Capabilities,* 13(4), pp. 561-579.

NGETAR, N.S., 2011. *Causes of Wetland Erosion at Craigieburn, Mpumalanga Province, South Africa*, PhD Thesis. Durban: School of Environmental Sciences, University of KwaZulu-Natal.

NGUBANE, S., 2016. *Assessing Spatial and Temporal Variations in Water Quality of the Upper uMngeni Catchment, KwaZulu – Natal, South Africa: 1989 – 2015.* MSc Thesis. Pietermaritzburg: School of Agricultural, Earth and Environmental Science, University of KwaZulu-Natal.

NICHOLS, P., LUCKE, T., DRAPPER, D. and WALKER, C., 2016. Performance evaluation of a floating treatment wetland in an urban catchment. *Water,* 8(6), pp. 1-8.

NICHOLSON, W. and SNYDER, C., 2012. *Microeconomic Theory: Basic Principles and Extensions.* 11 edn. Mason, OH: South-Western Cengage Learning.

NORDHAUS, W.D., 2017. Revisiting the social cost of carbon. *PNAS,* 114(7), pp. 1518-1523.

NORTON, B.G. and NOONAN, D., 2007. Ecology and valuation: Big changes needed. *Ecological Economics,* 63(4), pp. 664-675.

NORTON, B.G., 1984. Environmental ethics and weak anthropocentrism. *Environmental Ethics,* 6(2), pp. 131-148.
NTSHOTSHO, P., REYERS, B. and ESLER, K.J., 2011. Assessing the evidence base for restoration in South Africa. *Restoration Ecology,* 19(5), pp. 578-586.
O'NEILL, J., HOLLAND, A. and LIGHT, A., 2008. *Environmental Values.* 1 edn. Abingdon, Oxon: Routledge.
OBERHOLSTER, P.J. and ASHTON, P., 2008. *State of the Nation: An Overview of the Current Status of Water Quality and Eutrophication in South African Rivers and Reservoirs.* Report No. CSIR/NRE/WR/IR/2008/0075/C. Pretoria: Council for Scientific and Industrial Research.
OKEY, B.W., 1996. Systems approaches and properties, and agroecosystem health. *Journal of Environmental Management*, 48(2), pp. 187-199.
OLLIS, D.J., SNADDON, C.D., JOB, N.M. and MBONA, N., 2013. *Classification System for Wetlands and other Aquatic Ecosystems in South Africa. User Manual: Inland Systems.* Pretoria: South African National Biodiversity Institute.
OLSSON, P., FOLKE, C. and HAHN, T., 2004. Social-ecological transformation for ecosystem management: the development of adaptive co-management of a wetland landscape in southern Sweden. *Ecology and Society,* 9(4), pp. 2 [online].
O'NEILL, J. and SPASH, C.L., 2000. Conceptions of value in environmental decision-making. *Environmental Values,* 9(4), pp. 521-535.
OSTROM, E., 2007. A diagnostic approach for going beyond panaceas. *PNAS,* 104(39), pp. 15181-15187.
OSTROM, E., 2009. A General framework for analyzing sustainability of social-ecological systems. *Science,* 325(5939), pp. 419-422.
PAAVOLA, J. and BROMLEY, D.W., 2002. Contested choices. In: D.W. BROMLEY, ed, *Economics, Ethics, and Environmental Policy: Contested Choices.* Oxford: Blackwell Publishers Ltd, pp. 3-14.
PAGIOLA, S., VON RITTER, K. and BISHOP, J., 2004. *Assessing the economic value of ecosystem conservation.* Washington, D.C.: The World Bank.
PALMER, M., ALLAN, J.D., MEYER, J. and BERNHARDT, E.S., 2007. River restoration in the twenty-first century: Data and experiential knowledge to inform future efforts. *Restoration Ecology,* 15(3), pp. 472-481.
PALMER, M.A. and FILOSO, S., 2009. Restoration of ecosystem services for environmental markets. *Science,* 325(5940), pp. 575-576.
PALMER, R.W., TURPIE, J., MARNEWICK, G.C. and BATCHELOR, A.L., 2002. *Ecological and Economic Evaluation of Wetlands in the Upper Olifants River catchment, South Africa.* WRC Report No. 1162/1/02. Pretoria: Water Research Commission.
PAPANDREOU, A.A., 2003. Externality, convexity and institutions. *Economics and Philosophy,* 19(2), pp. 281-309.
PARKS, S. and GOWDY, J., 2013. What have economists learned about valuing nature? A review essay. *Ecosystem Services,* 3, pp. [online] e1-e10.
PASCUAL, U., MURADIAN, R., BRANDER, L., et al., 2010. The economics of valuing ecosystem services and biodiversity. In: P. KUMAR, ed, *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations.* London: Earthscan, pp. Chapter 5.
PATTISON, J., BOXALL, P.C. and ADAMOWICZ, W.L., 2011. The economic benefits of wetland retention and Restoration in Manitoba. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie,* 59(2), pp. 223-244.
PAVLINERI, N., SKOULIKIDIS, N.T. and TSIHRINTZIS, V.A., 2017. Constructed floating wetlands: A review of research, design, operation and management aspects, and data meta-analysis*.* *Chemical Engineering Journal,* 308, pp. 1120-1132.
PEARCE, D. and MORAN, D., 1994. *The Economic Value of Biodiversity.* London: Earthscan.
PEARCE, D., 1976. The limits of cost-benefit analysis as a guide to environmental policy. *Kyklos,* 29(1), pp. 97.

PEARCE, D.W. and SECCOMBE-HETT, T., 2000. Economic valuation and environmental decision-making in Europe. *Environmental Science and Technology,* 38(4), pp. 1419-1425.

PEH, K.S.H., BALMFORD, A., FIELD, R.H., et al., 2014. Benefits and costs of ecological restoration: Rapid assessment of changing ecosystem service values at a UK wetland. *Ecology and Evolution,* 4(20), pp. 3875-3886.

PELENC, J. and BALLET, J., 2015. Strong sustainability, critical natural capital and the capability approach. *Ecological Economics*, 112: pp. 36-44.

PENDLETON, L. and BALDERA, A., 2010. *Measuring and monitoring the economic effects of habitat restoration: A summary of a NOAA Blue Ribbon Panel.* Washington, D.C.: National Oceanic and Atmospheric Administration.

PERKINS, P.E., 2007. Feminist ecological economics and sustainability. *Journal of Bioeconomics,* 9(3), pp. 227-244.

PHILLIPS, T. and MADLOKAZI, N., 2011. *An Impact Assessment of the Research Funded by WRC on Wetland Management in South Africa.* WRC Report No. KV 253/10. Pretoria: Water Research Commission.

PICKERING, C. and BYRNE, J., 2014. The benefits of publishing systematic quantitative literature reviews for PhD candidates and other early-career researchers. *Higher Education Research & Development,* 33(3), pp. 534-548.

PIETERSE, C., 2018. Paddling a sewer*. The Witness*. [Online] Available: https://www.news24.com/SouthAfrica/News/paddling-in-a-sewer-20181107 8 November.

PILLAY, K., 2019b. Midmar mess 'is a crime'*. The Witness*. [Online] Available: https://www.news24.com/witness/news/midmar-mess-is-a-crime-20191125 26 November.

PILLAY, K., 2019a. Sewage dirties dam*. The Witness*. [Online] Available: https://www.news24.com/witness/news/sewage-dirties-dam-20191119 20 November.

POLASKY, S. and SEGERSON, K., 2009. Integrating ecology and economics in the study of ecosystem services: Some lessons learned. *Annual Review of Resource Economics,* 1(1), pp. 409-434.

POLLARD, S., BIGGS, H. and DU TOIT, D.R., 2014. A systemic framework for context-based decision making in natural resource management: reflections on an integrative assessment of water and livelihood security outcomes following policy reform in South Africa. *Ecology and Society,* 19(2), pp. 63.

POLLARD, S., KOTSE, D., ELLERY, W., et al., 2005. *Linking Water and Livelihoods: The Development of an Integrated Wetland Rehabilitation Plan in the Communal Areas of the Sand River Catchment as a Test Case.* Acornhoek, South Africa: Association for Water and Rural Development.

POLLARD, S., KOTZE, D.C. and FERRARI, G., 2008. *WET—OutcomeEvaluate Part 3B: Valuation of the Livelihood Benefits of Structural Rehabilitation Interventions in the Manalana Wetland.* WRC Report No. TT 343/08. Pretoria: Water Research Commission.

POTSCHIN, M.B. and HAINES-YOUNG, R., 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography: Earth and Environment,* 35(5), pp. 575-594.

POTTS, J., 2010. Ontology in economics. In: R. POLI and J. SEIBT, eds, *Theory and Applications of Ontology: Philosophical Perspectives.* Dortdrecht: Springer, pp. 277-286.

PRATO, T. and HEY, D., 2006. Economic analysis of wetland restoration along the Illinois River. *Journal of the American Water Resources Association,* 42(1), pp. 125-131.

PRATO, T., 1998. *Natural Resource and Environmental Economics.* 1 edn. Ames, Iowa: Iowa State University Press.

PRETTY, J.N., MASON, C.F., NEDWELL, D.B., HINE, R.E., LEAF, S. and DILS, R., 2003. Environmental costs of freshwater eutrophication in England and Wales. *Environmental Science & Technology,* 37(2), pp. 201-208.

PRIMMER, E., JOKINEN, P., BLICHARSKA, M., et al. 2015. Governance of ecosystem services: A framework for empirical analysis. *Ecosystem Services,* 16, pp. 158-166.

PRITCHARD, L., FOLKE, C. and GUNDERSON, L., 2000. Valuation of ecosystem services in institutional context. *Ecosystems,* 3(1), pp. 36-40.

RAMSAR CONVENTION SECRETARIAT (RAMSAR), 1971. *Final Act Annex I Convention on Wetlands of International Importance Especially as Waterfowl Habitat (original version).* Final Text adopted by the International Conference on the Wetlands and Waterfowl at Ramsar, Iran, 2 February. Gland: Ramsar Convention Secretariat.

RAMSAR CONVENTION SECRETARIAT (RAMSAR), 2002. *Principles and Guidelines for Wetland Restoration.* Gland: Ramsar Convention Secretariat.

RAMSAR CONVENTION SECRETARIAT (RAMSAR), 2010. *Addressing Change in Wetland Ecological Character: Addressing Change in the Ecological Character of Ramsar Sites and Other Wetlands.* Ramsar handbooks for the wise use of wetlands, 4th edition, vol. 19. Gland: Ramsar Convention Secretariat.

RAMSAR CONVENTION SECRETARIAT (RAMSAR), 2016. *An Introduction to the Convention on Wetlands (previously The Ramsar Convention Manual).* 5 edn. Gland, Switzerland: Ramsar Convention Secretariat.

RANGETI, I., 2014. *Determinants of Key Drivers for Potable Water Treatment Cost in uMngeni Basin*. PhD Thesis. Durban: Environmental Health in the Faculty of Health Sciences, University of Technology.

RASMUSSEN, L.V., MERTZ, O., CHRISTENSEN, A.E., et al., 2016. A combination of methods needed to assess the actual use of provisioning ecosystem services. *Ecosystem Services,* 17, pp. 75-86.

RAYMOND, C.M., SINGH, G.G., BENESSAIAH, K., et al., 2013. Ecosystem services and beyond: Using multiple metaphors to understand human–environment relationships. *Bioscience,* 63(7), pp. 536-546.

REBELO, L., MCCARTNEY, M.P. and FINLAYSON, C.M., 2010. Wetlands of Sub-Saharan Africa: Distribution and contribution of agriculture to livelihoods. *Wetlands Ecology and Management,* 18(5), pp. 557-572.

REY-BENAYAS, J.M., NEWTON, A.C., DIAZ, A. and BULLOCK, J.M., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-Analysis. *Science,* 325(5944), pp. 1121-1124.

REYERS, B., POLASKY, S., TALLIS, H., MOONEY, H.A. and LARIGAUDERIE, A., 2012. Finding common ground for biodiversity and ecosystem services. *Bioscience,* 62(5), pp. 503-507.

RIBOT, J.C. and PELUSO, N.L., 2003. A theory of access. *Rural Sociology,* 68(2), pp. 153-181.

RICKETTS, T.H. and LONSDORF, E., 2013. Mapping the margin: comparing marginal values of tropical forest remnants for pollination services. *Ecological Applications,* 23(5), pp. 1113-1123.

RIDDELL, E., 2011. *Characterisation of the Hydrological Processes and Responses to Rehabilitation of a Headwater Wetland of the Sand River, South Africa.* PhD Thesis*.* Pietermaritzburg: School of Bioresources Engineering and Environmental Hydrology, University of KwaZulu-Natal.

RIDDELL, E.S., LORENTZ, S.A. and KOTZE, D.C., 2012. The hydrodynamic response of a semi-arid headwater wetland to technical rehabilitation interventions. *Water SA,* 38(1), pp. 55-66.

RITTEL, H.W.J. and WEBBER, M.M., 1973. Dilemmas in a general theory of planning. *Policy Sciences,* 4(2), pp. 155-169.

RIVERS-MOORE, N.A., 2016. Exploratory use of a Bayesian network process for translating stakeholder perceptions of water quality problems in a catchment in South Africa. *Water SA,* 42(2), pp. 306-315.

ROBBINS, A.S.T. and DANIELS, J.M., 2012. Restoration and economics: A union waiting to happen? *Restoration Ecology,* 20(1), pp. 10-17.

ROBBINS, L., 1932. *An Essay on the Nature and Significance of Economic Science.* London: Macmillan and Co.

ROEBELING, P.C., CUNHA, M.C., ARROJA, L. and VAN GRIEKEN, M.E., 2011. Agricultural water pollution treatment for efficient water quality improvement in linked terrestrial and marine ecosystems. *Journal of Coastal Research,* 64, pp. 1936-1940.

ROHWER, Y. and MARRIS, E., 2016. Renaming restoration: conceptualizing and justifying the activity as a restoration of lost moral value rather than a return to a previous state. *Restoration Ecology,* 24(5), pp. 674-679.

ROLSTON, H., 1983. Values gone wild. *Inquiry,* 26(2), pp. 181-207.

RØPKE, I., 2004. The early history of modern ecological economics. *Ecological Economics,* 50(3–4), pp. 293-314.

RØPKE, I., 2005. Trends in the development of ecological economics from the late 1980s to the early 2000s. *Ecological Economics,* 55(2), pp. 262-290.

ROSENBERGER, R.S. and LOOMIS, J.B., 2003. Benefit transfer. In: P.A. CHAMP and K.J. BOYLE, eds, *A Primer on Nonmarket Valuation.* Dortdrecht: Springer, pp. 455-483.

ROSENBERGER, R.S. and STANLEY, T.D., 2006. Measurement, generalization, and publication: Sources of error in benefit transfers and their management. *Ecological Economics*, 60(2), pp. 372-378.

RUIZ-JAEN, M.C. and MITCHELL AIDE, T., 2005. Restoration success: How is it being measured? *Restoration Ecology,* 13(3), pp. 569-577.

RUSSELL, M. and GREENING, H., 2015. Estimating benefits in a recovering estuary: Tampa Bay, Florida. *Estuaries and Coasts,* 38, pp. S9-S18.

RUSSI, D., TEN BRINK, P., FARMER, A., BADURA, T., COATES, D., FÖRSTER, J., KUMAR, R. and DAVIDSON, N., 2013. *The Economics of Ecosystems and Biodiversity for Water and Wetlands.* London: The Institute for European Environmental Policy (IEEP) and Ramsar Secretariat.

RUTHERFURD, I.D., JERIE, K. and MARSH, N., 2000. *A Rehabilitation Manual for Australian Streams: Volume 1.* Canberra: Land and Water Resources Research and Development Corporation, Cooperative Research Centre for Catchment Hydrology.

SAGOFF, M., 2009. Intrinsic value: a reply to Justus et al. *Trends in Ecology & Evolution,* 24(12), pp. 643.

SAMSON, A. and MILES, R., 2016. Part 3: Selected behavioral science concepts. In: A. SAMSON, ed, *The Behavioral Economics Guide 2016.* [Online]: Behavioral Science Solutions Ltd, pp. 101-131.

SAMSON, A., 2014. *The Behavioral Economics Guide.* [Online]: Behavioral Science Solutions Ltd.

SATTERFIELD, T. and KALOF, L., 2005. Environmental values: An introduction - relativistic and axiomatic traditions in the study of environmental values. In: L. KALOF and T. SATTERFIELD, eds, *The Earthscan Reader in Environmental Values.* London: Earthscan, pp. xxi-xxxiii.

SCHNEIDER, F., KALLIS, G. and MARTINEZ-ALIER, J., 2010. Crisis or opportunity? Economic degrowth for social equity and ecological sustainability. Introduction to this special issue. *Journal of Cleaner Production*, 18(6), pp. 511-518.

SCHOON, M. and VAN DER LEEUW, S., 2015. The shift toward social-ecological systems perspectives: Insights into the human-nature relationship. *Natures Sciences Societes,* 32(2), pp. 166-174.

SCHULZ, C. and MARTIN-ORTEGA, J., 2018. Quantifying relational values — why not? *Current Opinion in Environmental Sustainability*, 35, pp. 15-21.

SCHUYT, K.D., 2005. Economic consequences of wetland degradation for local populations in Africa. *Ecological Economic,* 53(2), pp. 177-190.

SCOVRONICK, N. and TURPIE, J., 2010. *The Tourism Value of Nylsvley Floodplain.* WRC Report no. TT 441/09. Pretoria: Water Research Commission.

SEN, A.K., 1977. Rational fools: A critique of the behavioral foundations of economic theory. *Philosophy & Public Affairs,* 6(4), pp. 317-344.

SENT, E., 2006. Pluralisms in economics. In: S. KELLERT, H. LONGINO and K. WATERS, eds, *Scientific Pluralism.* Minneapolis: University of Minneapolis Press, pp. 80–101.

SHABMAN, L.A. and BATIE, S.S., 1978. Economic value of natural Coastal wetlands: A critique. *Coastal Zone Management Journal,* 4(3), pp. 231-247.

SHACKLETON, C., SHACKLETON, S., GAMBIZA, J., et al., 2008. *Links between Ecosystem Services and Poverty Alleviation: Situation Analysis for Arid and Semi-Arid Lands in Southern Africa.* Report to Ecosystem Services and Poverty Reduction Research Programme: DFID, NERC, ESRC. [Online]

Available: https://www.espa.ac.uk/projects/situation-analyses-projects , accessed 13/11/2021: Consortium on Ecosystems and Poverty in Sub-Saharan Africa (CEPSA).
SHACKLETON, C.M., SHACKLETON, S.E., NETSHILUVHI, T.R. and MATHABELA, F.R., 2005. The contribution and direct-use value of livestock to rural livelihoods in the Sand River catchment, South Africa. *African Journal of Range & Forage Science,* 22(2), pp. 127-140.
SOCIETY FOR ECOLOGICAL RESTORATION (SER), 2004. *The SER International Primer on Ecological Restoration.* Tucson: Society for Ecological Restoration International.
SOCIETY FOR ECOLOGICAL RESTORATION (SER), 2017. *SER 2017-2020 Strategic Plan.* Washington, D.C.: Society for Ecological Restoration International.
SÖDERBAUM, P., 1990. Neoclassical and institutional approaches to environmental economics. *Journal of Economic Issues,* 24(2), pp. 481-492.
SOUTH AFRICAN NATIONAL GOVERNMENT, 1998. *South African National Water Act (Act No. 36 of 1998).* Pretoria: South African National Government.
SPANGENBERG, J.H. and SETTELE, J., 2016. Value pluralism and economic valuation – defendable if well done. *Ecosystem Services,* 18, pp. 100-109.
SPANGENBERG, J.H., GÖRG, C., TRUONG, D.T., TEKKEN, V., BUSTAMANTE, J.V. and SETTELE, J., 2014. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. *International Journal of Biodiversity Science, Ecosystem Services & Management,* 10(1), pp. 40-53.
SPANGENBERG, J.H., VON HAAREN, C. and SETTELE, J., 2014. The ecosystem service cascade: Further developing the metaphor. Integrating societal processes to accommodate social processes and planning, and the case of bioenergy. *Ecological Economics,* 104, pp. 22-32.
SPASH, C.L. and ASLAKSEN, I., 2015. Re-establishing an ecological discourse in the policy debate over how to value ecosystems and biodiversity. *Journal of environmental management,* 159, pp. 245-253.
SPASH, C.L. and RYAN, A., 2012. Economic schools of thought on the environment: Investigating unity and division. *Cambridge Journal of Economics,* pp. 1-31.
SPASH, C.L., 1993. Economics, ethics, and long-term environmental. *Environmental Ethics,* 15(2), pp. 117-132.
SPASH, C.L., 1997. Ethics and environmental attitudes with implications for economic valuation. *Journal of Environmental Management*, 50(4), pp. 403-416.
SPASH, C.L., 1999. The development of environmental thinking in economics. *Environmental Values,* 8(4), pp. 413-435.
SPASH, C.L., 2008. Deliberative monetary valuation and the evidence for a new value theory. *Land Economics,* 84(3), pp. 469-488.
SPASH, C.L., 2013. The shallow or the deep ecological economics movement? *Ecological Economics,* 93, pp. 351-362.
SPENCER, P., PERKINS, P.E. and ERICKSON, J.D., 2018. Re-establishing Justice as a pillar of ecological economics through feminist perspectives. *Ecological Economics,* 152, pp. 191-198.
STATISTICS SOUTH AFRICA (STATS SA), 2012. *Census 2011 Municipal report – Mpumalanga.* Pretoria: Statistics South Africa.
STATISTICS SOUTH AFRICA (STATS SA), 2017. *Poverty Trends in South Africa: An Examination of Absolute Poverty between 2006 and 2015.* 03-10-06. Pretoria: Statistics South Africa.
STENSEKE, M., 2018. Connecting 'relational values' and relational landscape approaches. *Current Opinion in Environmental Sustainability,* 35, pp. 1-27.
STONE-JOVICICH, S., 2015. Probing the interfaces between the social sciences and social-ecological resilience; insights from integrative and hybrid perspectives in the social sciences. *Ecology and Society,* 20(2), pp. 25-48.
STREEVER, W.J., 1997. Trends in Australian wetland rehabilitation. *Wetlands Ecology and Management,* 5, pp. 5-18.

STUHR, J.J., 2003. Pragmatism about values and the valuable: Commentary on 'A pragmatic view on values in economics'. *Journal of Economic Methodology,* 10(2), pp. 213-221.
SUDING, K.N., 2011. Toward an era of restoration in ecology: Successes, failures, and opportunities ahead. *Annual Review of Ecology, Evolution, and Systematics,* 42, pp. 465-487.
SUTHERLAND, C., JEWITT, G., RISKO, S. and ET AL, 2019. *Water Resource Connectivity, Inter-Dependency and Social Relations from A Landscape Perspective.* Project K5/2354 Deliverable 14. Pietermaritzburg: Centre for Water Resources Research and School of Built Environment and Development Studies University of KwaZulu-Natal.
TEMESGEN, M. and RETTA, N., 2015. Nutritional potential, health and food security benefits of *Taro Colocasia Esculenta (L.)*: A Review. *Food Science and Quality Management,* 36, pp. 23-30.
TERRY, S., 2017. Water and Environment Scientist. Pietermaritzburg: Umgeni Water. Personal Communication, 26 April.
THE WITNESS, 2015. Umgeni Water: Midmar Dam water safe. *The Witness*. [Online] Available: https://www.news24.com/witness/news/umgeni-water-midmar-dam-water-safe-20151123 24 November.
THORNTON, J.A., HARDING, W.R., DENT, M., et al., 2013. Eutrophication as a 'wicked' problem. *Lakes & Reservoirs: Research & Management,* 18(4), pp. 298-316.
THROSBY, D., 1999. Cultural capital. *Journal of Cultural Economics,* 23(1), pp. 3-12.
TONG, C., FEAGIN, R.A., LU, J., ZHANG, X., ZHU, X., WANG, W. and HE, W., 2007. Ecosystem service values and restoration in the urban Sanyang wetland of Wenzhou, China. *Ecological Engineering,* 29(3), pp. 249-258.
TOURISM KWAZULU-NATAL, 2019. *Midmar Mile 2019: Economic Impact Assessment.* Durban [Online] Available: https://zulu.org.za/corporate/event-impact-assessments/ , accessed 03/11/2020: Department of Economic Development, Tourism and Environmental Affairs.
TURNER, K.G., ANDERSON, S., GONZALES-CHANG, M., et al., 2016. A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration. *Ecological Modelling*, 319, pp. 190-207.
TURNER, R.K., GEORGIOU, S. and FISHER, B., 2008. *Valuing Ecosystem Services: The Case of Multi-functional Wetlands.* London: Earthscan.
TURNER, R.K., PAAVOLA, J., COOPER, P., et al., 2003. Valuing nature: Lessons learned and future research directions. *Ecological Economics,* 46(3), pp. 493-510.
TURNER, R.K., VAN DEN BERGH, J.C.J.M., SÖDERQVIST, T., et al., 2000. Ecological-economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics,* 35(1), pp. 7-23.
TURPIE, J. and KLEYNHANS, M., 2010. *Wetland Valuaton, Volume 4: A Protocol for the Quantification and Valuation of Wetland Ecosystem Services.* WRC Report No. TT 443/09. Pretoria: Water Research Commission.
TURPIE, J., DAY, E., ROSS-GILLESPIE, V. and LOUW, A., 2010. *Estimation of the Water Quality Amelioration Value of Wetlands: A Case Study of the Western Cape, South Africa. Environment for Development Discussion Paper Series. .* EfD DP 10-15. Cape Town: Environment for Development and Environmental Policy Research Unit, University of Cape Town.
TURPIE, J., JOUBERT, A., BABIKER, H. et al., 2009. *Integrated Ecological-Economic Modelling As an Estuarine Management Tool: A Case Study of the East Kleinemonde Estuary. Volume I: The Economic Value of the East Kleinemonde Estuary and Impacts of Changes in Freshwater Inputs.* WRC Report No. 1679/1/08. Pretoria: Water Research Commission.
TURPIE, J., SMITH, B., EMERTON, L. and BARNES, J., 1999. *Economic Value of the Zambezi Basin Wetlands.* Pretoria: IUCN Regional Office for Southern Africa.
TURPIE, J.K. and JOUBERT, A., 2001. Estimating potential impacts of a change in river quality on the tourism value, Kruger National Park: An application of travel cost, contingent and conjoint valuation methods. *Water SA,* 27, pp. 387-398.
TURPIE, J.K., FORSYTHE, K.J., KNOWLES, A., BLIGNAUT, J. and LETLEY, G., 2017. Mapping and valuation of South Africa's ecosystem services: A local perspective. *Ecosystem Services,* 27, pp. 179-192.

UMGENI WATER, 2013. *Infrastructure Master Plan 2013/2014 – 2043/2044, Volume 2.* Pietermaritzburg: Umgeni Water.

UMGENI WATER, 2018. *Annual Report 2017/2018.* Pietermaritzburg: Umgeni Water.

UMGENI WATER, 2019. *Infrastructure Master Plan 2019: 2019/2020 – 2049/2050, Volume 2: Mgeni System.* Pietermaritzburg: Umgeni Water.

UNITED NATIONS ENVIRONMENT PROGRAMME (UNEP) & FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS (FAO), 2019. *United Nations Decade on Ecosystem Restoration (2021–2030).* Nairobi: United Nations Environment Programme.

UNITED NATIONS ENVIRONMENT PROGRAMME (UNEP), 2015. *Economic Valuation of Wastewater: The Cost of Action and the Cost of No Action.* Nairobi: United Nations Environment Programme.

VAN DEN BERGH, J.C.J.M., FERRER-I-CARBONELL, A. and MUNDA, G., 2000. Alternative models of individual behaviour and implications for environmental policy. *Ecological Economics*, 32(1), pp. 43-61.

VAN DEVENTER, R., 2012. *Impact of Land Use on Water Quality and Aquatic Ecosystem Health of Stream Networks in the Upper uMngeni Catchment Feeding Midmar Dam, KwaZulu-Natal, South Africa.* MSc Thesis. Pietermaritzburg: School of Agricultural, Earth and Environmental Sciences, University of KwaZulu-Natal.

VAN GINKEL, C.E., 2011. Eutrophication: present reality and future challenges for South Africa. *Water SA,* 37(5), pp. 693-701.

VAN KERKHOFF, L. and BERRY, H., 2016. Serving the public good: Empirical links between governance and research investment in the context of global environmental change. *Ecological Economics,* 125, pp. 101-107.

VAN ZYL, H., LEMAN, A. and JANSEN, A., 2004. *The Costs and Benefits of Urban River and Wetland Rehabilitation Projects with Specific Reference to Their Implications for Municipal Finance: Case Studies in Cape Town.* WRC Report No: KV159/04. Pretoria: Water Research Commission.

VATN, A. and BROMLEY, D.W., 1994. Choices without prices without apologies. *Journal of Environmental Economics and Management,* 26(2), pp. 129-148.

VATN, A., 2005. *Institutions and the Environment.* Cheltenham: Edward Elgar.

VATN, A., 2009. An institutional analysis of methods for environmental appraisal. *Ecological Economics,* 68(8–9), pp. 2207-2215.

VATN, A., 2010. An institutional analysis of payments for environmental services. *Ecological Economics,* 69(6), pp. 1245-1252.

VENKATACHALAM, L., 2007. Environmental economics and ecological economics: Where they can converge? *Ecological Economics,* 61(2–3), pp. 550-558.

VENKATACHALAM, L., 2008. Behavioral economics for environmental policy. *Ecological Economics,* 67(4), pp. 640-645.

VERDONE, M., 2015. *A Cost-Benefit Framework for Analyzing Forest Landscape Restoration Decisions.* Gland: International Union for Conservation of Nature (IUCN).

VERHOEVEN, J.T.A., ARHEIMER, B., YIN, C. and HEFTING, M.M., 2006. Regional and global concerns over wetlands and water quality. *Trends in Ecology & Evolution,* 21(2), pp. 96-103.

VIEIRA DA SILVA, L., EVERARD, M. and SHORE, R.G., 2014. Ecosystem services assessment at Steart Peninsula, Somerset, UK. *Ecosystem Services,* 10, pp. 19-34.

VOGEL, C., SCOTT, D., CULWICK, C.E. and SUTHERLAND, C., 2016. Environmental problem-solving in South Africa: Harnessing creative imaginaries to address 'wicked' challenges and opportunities. *South African Geographical Journal,* 98(3), pp. 515-530.

VUNDLA, T., BLIGNAUT, J.N. and CROOKES, D., 2017. Aquatic weeds: To control or not to control. The case of the Midmar Dam, KwaZulu-Natal, South Africa. *African Journal of Agricultural and Resource Economics,* 12(4), pp. 412-429.

WAINGER, L. and MAZZOTTA, M., 2011. Realizing the potential of ecosystem services: A framework for relating ecological changes to economic benefits. *Environmental Management,* 48(4), pp. 710.

WAINGER, L., KING, D., SALZMAN, J. and BOYD, J., 2001. Wetland value indicators for scoring mitigation trades. *Stanford Environmental Law Journal,* 20(2), pp. 413-478.
WAITE, R., BURKE, L., GRAY, E., et al., 2014. *Coastal Capital: Ecosystem Valuation for Decision Making in the Caribbean.* Washington, D.C.: World Resources Institute.
WALTERS, D. and BROWNE, M., 2018. *An Integrated Report Detailing the Monitoring and Evaluation of the Rehabilitation Undertaken At the Greater Edendale Mall and Manalana Wetland Sites: Manalana Wetland.* Project K5/2344 Deliverable 9. Hilton, South Africa: GroundTruth Water, Wetlands, Biodiversity and Environmental Engineering.
WALTERS, D., KOTZE, D., COWDEN, C., et al., 2019. *WET-RehabEvaluate Version 2: An Integrated Monitoring and Evaluation Framework to Assess Wetland Rehabilitation in South Africa.* WRC Report No. 2344/1/19. Pretoria: Water Research Commission.
WANG, Z., FEI, X., HE, S., HUANG, J. and ZHOU, W., 2017. Comparison of heterotrophic and autotrophic denitrification processes for treating nitrate-contaminated surface water. *Science of the Total Environment,* 579, pp. 1706-1714.
WATER RESEARCH COMMISSION (WRC), 2002. *State-of-Rivers Report: uMngeni River and Neighbouring Rivers and Streams.* WRC Report No.TT 200/02. Pretoria: Water Research Commission.
WEBER, M.A. and STEWART, S., 2009. Public values for river restoration options on the Middle Rio Grande. *Restoration Ecology,* 17(6), pp. 762-771.
WESLEY, E. and PETERSON, F., 1993. Time preference, the environment and the interests of future generations. *Journal of Agricultural and Environmental Ethics,* 6(2), pp. 107-126.
WESTERBERG, V.H., LIFRAN, R. and OLSEN, S.B., 2010. To restore or not? A valuation of social and ecological functions of the Marais des Baux wetland in Southern France. *Ecological Economics,* 69(12), pp. 2383-2393.
WESTMAN, W.E., 1977. How much are nature's services worth? *Science,* 197(4307), pp. 960-964.
WESTON, A., 1985. Beyond intrinsic value: Pragmatism in environmental ethics. *Environmental Ethics,* 7(4), pp. 321-339.
WESTON, A., 2009. *The Incompleat Eco Philosopher.* Albany: State University of New York Press.
WHITEHEAD, J.C., PATTANAYAK, S.K., VAN HOUTVEN, G.L. and GELSO, B.R., 2008. Combining revealed and stated preference data to estimate the nonmarket value of ecological services: An assessment of the state of the science. *Journal of Economic Surveys,* 22(5), pp. 872-908.
WIELAND, R., RAVENSBERGEN, S., GREGR, E.J., SATTERFIELD, T. and CHAN, K.M.A., 2016. Debunking trickle-down ecosystem services: The fallacy of omnipotent, homogeneous beneficiaries. *Ecological Economics,* 121, pp. 175-180.
WIGHT, J., 2012. Ethics and critical thinking. In: G.M. HOYT and K. MCGOLDRICK, eds, *International Handbook on Teaching and Learning Economics.* Cheltenham: Edward Elgar, pp. 197-204.
WILDEMUTH, B.M., 1993. Post-positivist research: Two examples of methodological pluralism. *The Library Quarterly,* 63(4), pp. 450-468.
WILLIS, J., 2007. *Foundations of Qualitative Research.* California: Sage Publications, Inc.
WOLFF, R.D., 2013. Alternatives to Capitalism. *Critical Sociology,* 39(4), pp. 487-490.
WOLFF, S., SCHULP, C.J.E. and VERBURG, P.H., 2015. Mapping ecosystem services demand: A review of current research and future perspectives. *Ecological Indicators,* 55, pp. 159-171.
WOODWARD, R.T. and WUI, Y., 2001. The economic value of wetland services: A meta-analysis. *Ecological Economics,* 37(2), pp. 257-270.
WORTLEY, L., HERO, J. and HOWES, M., 2013. Evaluating ecological restoration success: A Review of the literature. *Restoration Ecology,* 21(5), pp. 537-543.
YOUNG, R.A., 1996. *Measuring Economic Benefits for Water Investments and Policies.* World Bank Technical Paper No. 338. Washington, D.C.: The World Bank.
ZHOU, W., SUN, Y., WU, B., et al., 2011. Autotrophic denitrification for nitrate and nitrite removal using sulfur-limestone. *Journal of Environmental Sciences*, 23(11), pp. 1761-1769.

ZHUANG, J., LIANG, Z., LIN, T. and DE GUZMAN, F., 2007. *Theory and Practice in the Choice of Social Discount Rate for Cost-Benefit Analysis: A Survey.* ERD Working Paper No. 94. Mandaluyong City: Asian Development Bank.

ZIMMERMAN, M.J., 2015. Intrinsic vs. extrinsic value. In: E.N. ZALTA, ed, *The Stanford Encyclopedia of Philosophy (Spring 2015 Edition).* [Online] Available: http://plato.stanford.edu/archives/spr2015/entries/value-intrinsic-extrinsic/ : Metaphysics Research Lab, Stanford University.

APPENDICES

Appendix 2.1: Ecosystem valuation frameworks and guidelines

Framework	Reference	Publication
An appraisal framework for wetland valuation	Barbier et al. (1997)	Ramsar Convention report
Phases of a joint ecosystem assessment and economic analysis for a single scenario	Bateman et al. (2011)	Environmental and Resource Economics Journal article
Benefit valuation steps	Brouwer & Georgiou (2012)	WHO report
A Framework for wetland valuation	de Groot et al. (2006)	Ramsar Convention report
The 'Impact Pathway' approach to the valuation of ecosystem services	DEFRA (2011)	UK Government report
Stages in economic and biodiversity assessment of wetlands	Emerton and Kekulandala (2003)	IUCN report
An integrated and expanded approach to ecological valuation	EPA (2009)	USA Government report
Aquatic ecosystem service evaluation framework	Ginsburg et al. (2010)	SA Water Research Commission report
The economic approach to valuation	Heal et al. (2005)	National Academies Press
The ecosystem valuation framework	Hein et al. (2006)	Ecological Economics Journal article
Template for the assessment and valuation of water quality-related services	Keeler et al. (2012)	*PNAS* article
Practical steps in the execution of cost-benefit analysis	Mullins et al. (2014)	SA Water Research Commission report
Integrating ecology and economics in the study of ecosystem services: Some Lessons Learned	Polasky and Sergerson (2009)	Annual Review of Resource Economics Journal article
A stepwise approach to assessing nature's benefits	TEEB (2010)	TEEB report
A protocol for the quantification and valuation of wetland ecosystem services	Turpie & Kleynhans (2010)	SA Water Research Commission report
The ecosystem services approach: Valuation of multi-functional wetlands	Turner et al. (2008)	Book, published by EarthScan
Methodological framework of Steart ecosystem services assessment study	Vieira et al. (2014)	Ecosystem Services Journal article
Steps in conducting coastal ecosystem valuation to inform decision making in the Caribbean	Waite et al. (2014)	World Resources Institute report

Note: Highlighted references refer to South African publications.

Source: Authors own review.

Appendix 3.1: Summary of literature reviews related to economics, restoration and wetlands

Study	Search terms	Results	Study conclusions
Reviews related to the economics of wetland restoration			
Ghermandi et al. (2010). Values of natural and human-made wetlands, A meta-analysis	Not given, expands an earlier database of one of the authors (Brander et al., 2006)	170 valuation studies, 68 journal articles, of which 3 address wetland restoration, 8 address human-made wetlands	The number of studies addressing the economic values of human-made wetlands is rather limited; Geographical distribution of valued sites is skewed toward temperate Northern latitudes and equatorial regions
Reviews related to the economics of ecosystem restoration			
Aronson et al. (2010). Are Socioeconomic Benefits of Restoration Adequately Quantified?	"restoration" or "rehabilitation"	1582 (peer reviewed)	Aquatic systems are underrepresented; Socio-economic contributions of restoration to society are underemphasized, or often ignored altogether
Robbins and Daniels (2012). Restoration and Economics, A Union Waiting to Happen	97 (including 31 non-journal reports)		Costs are rarely discussed or analysed in the literature
Blignaut et al. (2013). Establishing the links between economic development and the restoration of natural capital	"restoration" or "rehabilitation"	1582 (peer reviewed), 513 considered rivers and wetlands	Further research is needed to assess and quantify the likely or known changes in ecosystem service generation and delivery and to assess and quantify the benefits
de Groot et al. (2013). Benefits of investing in ecosystem restoration	*Draws from the database of Aronson et al. (2010) and Blignaut et al. (2013).*	Costs – 94 studies Benefits – 225 studies	Costs of restoration are less frequently reported or analysed, compared to benefits. Detailed studies that monitor costs and benefits of restoration over time are needed.
Blignaut et al. (2014). The economics of restoration, looking back and leaping forward	"restore" or "restoration;" and "economy" or "economics;" and "ecology" or "environment" or "ecosystem"	650 (journal articles)	The economics of restoration is a subject rapidly gaining ground
Reviews related to ecosystem restoration			
Ruiz-Jaen and Aide (2005). Restoration Success. How Is It Being Measured?	All articles published in Restoration Ecology Vols. 1[1]–11[4], only articles that used seeding or planting techniques to assist the restoration process and whose main objective was to restore a site or evaluate restoration success were considered	68 studies	Wetlands were the habitat most frequently (19%) studied; There was no mention of economic impact or value produced

Study	Search terms	Results	Study conclusions
Ntshotsho et al. (2011). Assessing the evidence base for restoration in South Africa	(ecological, post-mining, and rangeland), (restoration, rehabilitation, and revegetation) (project/s and program/s). The phrase "South Africa" was used as a suffix in all the search combinations	10 restoration programs assessed	Socio-economic indicators were measured more regularly than ecological indicators; Outcome-based indicators were lacking (there was no mention of measuring or monitoring the economic value produced)
Wortley et al. (2013). Evaluating Ecological Restoration Success, A Review	(restoration or restored) and (eco*) and (monitor* or success* or evaluate* or assess*)	301 (primary academic literature), limited to terrestrial restoration	3.9 % of studies considered economic attributes, but the review 'found no economic measure of ecosystem services in post-implementation evaluations'
Meli et al. (2014). Restoration Enhances Wetland Biodiversity and Ecosystem Service Supply, but Results Are Context-Dependent: A Meta-Analysis	(riparian OR river* OR lake OR mangroves OR marsh OR stream OR wetland) AND (restor* OR re-creat* OR rehabilitat* OR forest* OR reforest* OR afforest* OR plant* OR recover*) AND ((ecosystem OR environment) AND (service OR function*))	1931, 70 satisfied the chosen criteria and were included in the meta-analysis	No information on benefit valuation was reported; The review reported that none of the studies considered restoration costs

Appendix 3.2: List of references for the publications examined in the literature review (N=63 studies)

	Reference
1	Adame, M.F., Hermoso, V., Perhans, K., Lovelock, C.E. and Herrera-Silveira, J.A. 2015. Selecting cost-effective areas for restoration of ecosystem services. *Conservation Biology,* 29 (2):493-502.
2	Adusumilli, N. 2015. Valuation of ecosystem services from wetlands mitigation in the United States. *Land,* 4 (1):182-196.
3	Alfranca, O., García, J. and Varela, H. 2011. Economic valuation of a created wetland fed with treated wastewater located in a peri-urban park in Catalonia, Spain. *Water Science & Technology,* 63 (5):891-898.
4	Awondo, S.N., Egan, K.J. and Dwyer, D. 2011. Increasing beach recreation benefits by using wetlands to reduce contamination. *Marine Resource Economics,* 26 (1):1-15.
5	Bauer, D.M., Cyr, N.E. and Swallow, S.K. 2004. Public preferences for compensatory mitigation of salt marsh losses: a contingent choice of alternatives. *Conservation Biology,* 18 (2):401-411.
6	Bloczynski, J.A., Bogart, W.T., Hobbs, B.F. and Koonce, J.F. 2000. Irreversible investment in wetlands preservation: optimal ecosystem restoration under uncertainty. *Environmental management,* 26 (2):175-193.
7	Broekx, S., Smets, S., Liekens, I., Bulckaen, D. and De Nocker, L. 2011. Designing a long-term flood risk management plan for the Scheldt estuary using a risk-based approach. *Natural Hazards,* 57 (2):245-266.
8	Caffey, R.H., Wang, H. and Petrolia, D.R. 2014. Trajectory economics: Assessing the flow of ecosystem services from coastal restoration. *Ecological Economics,* 100:74-84.
9	Cardoch, L., Day Jr., J.W., Rybczyk, J.M. and Kemp, G.P. 2000. An economic analysis of using wetlands for treatment of shrimp processing wastewater — a case study in Dulac, LA. *Ecological Economics,* 33 (1):93-101.
10	Chen, Z.M., Chen, G.Q., Chen, B., Zhou, J.B., Yang, Z.F. and Zhou, Y. 2009. Net ecosystem services value of wetland: environmental economic account. *Communications in Nonlinear Science and Numerical Simulation,* 14 (6):2837-2843.
11	Dodds, W.K., Wilson, K.C., Rehmeier, R.L., Knight, G.L., Wiggam, S., Falke, J.A., Dalgleish, H.J. and Bertrand, K.N. 2008. Comparing ecosystem goods and services provided by restored and native lands. *Bioscience,* 58 (9):837-845.
12	Earnhart, D. 2001. Combining revealed and stated preference methods to value environmental amenities at residential locations. *Land Economics,* 77 (1):12-29.
13	Englehardt, J.D. 1998. Ecological and economic risk analysis of everglades: phase I restoration alternatives. *Risk Analysis,* 18 (6):755-771.
14	Gren, I., Söderqvist, T. and Wulff, F. 1997. Nutrient reductions to the Baltic Sea: ecology, costs and benefits. *Journal of environmental management,* 51 (2):123-143.
15	Grossmann, M. 2012. Economic value of the nutrient retention function of restored floodplain wetlands in the Elbe River basin. *Ecological Economics,* 83 (0):108-117.
16	Grossmann, M. and Dietrich, O. 2012. Social benefits and abatement costs of greenhouse gas emission reductions from restoring drained fen wetlands: a case study from the Elbe River basin (Germany). *Irrigation and Drainage,* 61 (5):691-704.
17	Gutrich, J.J. and Hitzhusen, F.J. 2004. Assessing the substitutability of mitigation wetlands for natural sites: estimating restoration lag costs of wetland mitigation. *Ecological Economics,* 48 (4):409-424.

18 He, J., Ai, J., Zhu, X. and Sun, X. 2015. Ecological compensation standards of wetland restoration projects. *Polish Journal of Environmental Studies,* 24 (6):2421-2432.

19 He, J., Sun, X. and Zhu, X. 2015. Spatial disparities of the willingness of the residents to pay for the wetland restoration of Taihu Lake and its integration into decision making: a case study on Wuxi, China. *Environmental monitoring and assessment,* 187 (8):492.

20 Humphries, A.T. and La Peyre, M.K. 2015. Oyster reef restoration supports increased nekton biomass and potential commercial fishery value. *PeerJ,* 3:e1111.

21 Hyman, J.B. and Leibowitz, S.G. 2000. A general framework for prioritizing land units for ecological protection and restoration. *Environmental management,* 25 (1):23-35.

22 Ito, N., Takeuchi, K., Kuriyama, K., Shoji, Y., Tsuge, T. and Mitani, Y. 2009. The influence of decision-making rules on individual preferences for ecological restoration: evidence from an experimental survey. *Ecological Economics,* 68 (8–9):2426-2431.

23 Jenkins, W.A., Murray, B.C., Kramer, R.A. and Faulkner, S.P. 2010. Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. *Ecological Economics,* 69:1051-1061.

24 Johnston, R.J., Magnusson, G., Mazzotta, M.J. and Opaluch, J.J. 2002. Combining economic and ecological indicators to prioritize salt marsh restoration actions. *American Journal of Agricultural Economics,* 84 (5, Proceedings Issue):1362-1370.

25 Johnston, R.J., Opaluch, J.J., Magnusson, G. and Mazzotta, M.J. 2005. Who are resource nonusers and what can they tell us about nonuse values? Decomposing user and nonuser willingness to pay for coastal wetland restoration. *Water Resources Research,* 41 (W07017).

26 Kaza, N. and BenDor, T.K. 2013. The land value impacts of wetland restoration. *Journal of Environmental Management,* 127:289-299.

27 Kim, T. and Petrolia, D.R. 2013. Public perceptions of wetland restoration benefits in Louisiana. *ICES Journal of Marine Science: Journal du Conseil,* 70 (5):1045-1054.

28 Kirk, J.A., Wise, W.R. and Delfino, J.J. 2004. Water budget and cost-effectiveness analysis of wetland restoration alternatives: a case study of Levy Prairie, Alachua County, Florida. *Ecological Engineering,* 22 (1):43-60.

29 Klimkowska, A., Dzierża, P., Brzezińska, K., Kotowski, W. and Mędrzycki, P. 2010. Can we balance the high costs of nature restoration with the method of topsoil removal? Case study from Poland. *Journal for Nature Conservation,* 18 (3):202-205.

30 Lant, C.L., Kraft, S.E. and Gillman, K.R. 1995. The 1990 farm bill and water quality in Corn Belt watersheds: Conserving remaining wetlands and restoring farmed wetlands. *Journal of Soil and Water Conservation,* 50 (2):201-205.

31 Lantz, V., Boxall, P., Kennedy, M. and Wilson, J. 2013. The valuation of wetland conservation in an urban/peri urban watershed. *Regional Environmental Change,* 13 (5):939-953.

32 Lei, L., Jian, L., Yutao, W., Nvjie, W. and Renqing, W. 2011. Cost-benefit analysis and payments for watershed-scale wetland rehabilitation: a case study in Shandong Province, China. *International Journal of Environmental Research,* 5 (3):787-796.

33 Loesch, C.R., Reynolds, R.E. and Hansen, L.T. 2012. An assessment of re-directing breeding waterfowl conservation relative to predictions of climate change. *Journal of Fish and Wildlife Management,* 3 (1):1-22.

34 Lundhede, T., Bille, T. and Hasler, B. 2013. Exploring preferences and non-use values for hidden archaeological artefacts: a case from Denmark. *International Journal of Cultural Policy,* 19 (4):501-530.

35 Martin, J.F. 2002. Emergy valuation of diversions of river water to marshes in the Mississippi River Delta. *Ecological Engineering,* 18 (3):265-286.

36 Mayer, A.L. 2001. The effect of limited options and policy interactions on water storage policy in South Florida. *Journal of environmental management,* 63 (1):87-102.
37 Merino, J., Aust, C. and Caffey, R. 2011. Cost-efficacy in wetland restoration projects in coastal Louisiana. *Wetlands,* 31 (2):367-375.
38 Meyerhoff, J. and Dehnhardt, A. 2007. The European Water Framework Directive and economic valuation of wetlands: the restoration of floodplains along the River Elbe. *European Environment,* 17 (1):18-36.
39 Milon, J.W. and Scrogin, D. 2006. Latent preferences and valuation of wetland ecosystem restoration. *Ecological Economics,* 56 (2):162-175.
40 Minello, T.J., Rozas, L.P., Caldwell, P.A. and Liese, C. 2012. A comparison of salt marsh construction costs with the value of exported shrimp production. *Wetlands,* 32 (5):791-799.
41 Morrison, M. 2002. Understanding local community preferences for wetland quality. *Ecological Management & Restoration,* 3 (2):127-134.
42 Pattison, J., Boxall, P.C. and Adamowicz, W.L. 2011. The economic benefits of wetland retention and restoration in Manitoba. *Canadian Journal of Agricultural Economics,* 59 (2):223-244.
43 Peh, K.S.H., Balmford, A., Field, R.H., Lamb, A., Birch, J.C., Bradbury, R.B., Brown, C., Butchart, S.H.M., Lester, M., Morrison, R., Sedgwick, I., Soans, C., Stattersfield, A.J., Stroh, P.A., Swetnam, R.D., Thomas, D.H.L., Walpole, M., Warrington, S. and Hughes, F.M.R. 2014. Benefits and costs of ecological restoration: rapid assessment of changing ecosystem service values at a UK wetland. *Ecology and Evolution,* 4 (20):3875-3886.
44 Petrolia, D.R., Interis, M.G. and Hwang, J. 2014. America's Wetland? A national survey of willingness to pay for restoration of Louisiana's coastal wetlands. *Marine Resource Economics,* 29 (1):17-37.
45 Prato, T. and Hey, D. 2006. Economic analysis of wetland restoration along the Illinois River. *Journal of the American Water Resources Association,* 42 (1):125-131.
46 Reddy, S.M.W., McDonald, R.I., Maas, A.S., Rogers, A., Girvetz, E.H., North, J., Molnar, J., Finley, T., Leathers, G. and DiMuro, J.L. 2015. Finding solutions to water scarcity: incorporating ecosystem service values into business planning at The Dow Chemical Company's Freeport, TX facility. *Ecosystem Services,* 12:94-107.
47 Roebeling, P.C., Cunha, M.C., Arroja, l. and van Grieken, M.E. 2011. Agricultural water pollution treatment for efficient water quality improvement in linked terrestrial and marine ecosystems. *Journal of Coastal Research,* SI 64:1936-1940.
48 Rönnbäck, P., Crona, B. and Ingwall, L. 2007. The return of ecosystem goods and services in replanted mangrove forests: perspectives from local communities in Kenya. *Environmental Conservation,* 34 (04):313-324.
49 Rozas, L.P., Caldwell, P. and Minello, T.J. 2005. The fishery value of salt marsh restoration projects. *Journal of Coastal Research, SI 40*:37-50.
50 Russell, M. and Greening, H. 2015. Estimating benefits in a recovering estuary: Tampa Bay, Florida. *Estuaries and Coasts,* 38:S9-S18.
51 Scemama, P. and Levrel, H. 2016. Using Habitat Equivalency Analysis to assess the cost effectiveness of restoration outcomes in four institutional contexts. *Environmental management,* 57 (1):109-122.
52 Shepherd, D., Burgess, D., Jickells, T., Andrews, J., Cave, R., Turner, R.K., Aldridge, J., Parker, E.R. and Young, E. 2007. Modelling the effects and economics of managed realignment on the cycling and storage of nutrients, carbon and sediments in the Blackwater estuary UK. *Estuarine, Coastal and Shelf Science,* 73 (3–4):355-367.
53 Shultz, S.D. and Leitch, J.A. 2003. The feasibility of restoring previously drained wetlands to reduce flood damage. *Journal of Soil and Water Conservation,* 58 (1):21-29.
54 Sparks, E.L., Cebrian, J., Biber, P.D., Sheehan, K.L. and Tobias, C.R. 2013. Cost-effectiveness of two small-scale salt marsh restoration designs. *Ecological Engineering,* 53:250-256.

55 Spash, C.L. 2000. Ecosystems, contingent valuation and ethics: the case of wetland re-creation. *Ecological Economics,* 34 (2):195-215.

56 Speight Jr., C.C.C. 2013. Lessons Learned From A Wetlands Restoration Project. *Cost Engineering, July/August 2013*:27-33.

57 Tolvanen, A., Juutinen, A. and Svento, R. 2013. Preferences of local people for the use of peatlands: the case of the richest peatland region in Finland. *Ecology and Society,* 18 (2).

58 Tong, C., Feagin, R.A., Lu, J., Zhang, X., Zhu, X., Wang, W. and He, W. 2007. Ecosystem service values and restoration in the urban Sanyang wetland of Wenzhou, China. *Ecological Engineering,* 29 (3):249-258.

59 Tri, H.N., Adger, W. and Kelly, P. 1998. Natural resource management in mitigating climate impacts: the example of mangrove restoration in Vietnam. *Global Environmental Change,* 8 (1):49-61.

60 Turner, R.E. and Warren, R.S. 2003. Valuation of continuous and intermittent Phragmites control. *Estuaries,* 26 (2):618-623.

61 Vieira da Silva, L., Everard, M. and Shore, R.G. 2014. Ecosystem services assessment at Steart Peninsula, Somerset, UK. *Ecosystem Services,* 10 (0):19-34.

62 Westerberg, V.H., Lifran, R. and Olsen, S.B. 2010. To restore or not? A valuation of social and ecological functions of the Marais des Baux wetland in Southern France. *Ecological Economics,* 69 (12):2383-2393.

63 Zilverberg, C.J., Johnson, W.C., Boe, A., Owens, V., Archer, D.W., Novotny, C., Volke, M. and Werner, B. 2014. Growing *Spartina pectinata* in previously farmed prairie wetlands for economic and ecological benefits. *Wetlands,* 34 (5):853-864.

Appendix 7.1: Defining elements of the framework

Definition	Indicators and examples
Wetland rehabilitation	
A process of assisting in a recovery of the condition or health of a wetland that has been degraded, or halting a decline in (maintaining) the health of a wetland that is in the process of degrading (Kotze et al., 2008). Implicit, is a difference in the health and functioning of the wetland between the 'with' and 'without' rehabilitation cases. The interpretation of wetland rehabilitation used in this framework is broad and includes the definition of (ecosystem) restoration, rehabilitation and repair as defined by McDonald et al. (2016). In the South African wetland management context, interventions in wetland ecosystems are generally referred to as 'rehabilitation' in recognition of the uncertainty in attaining some former state and, increasingly, the case of designing wetland systems towards a desired state (e.g., to promote the supply of a particular ecosystem service).	Typical rehabilitation activities include the construction of concrete and earth structures to redistribute water flows across the surface of the wetland and / or structures to control or halt erosion; re-shaping earthworks and the infilling of artificial drains; removal of solid waste (e.g., building rubble) and alien plant vegetation; and revegetation of disturbed areas. Example: structural interventions (concrete weir structures and earthen berms) to redistribute water flows across the surface of the wetland which had previously been largely confined to a single channel due to anthropogenic impacts including the excavation of artificial drainage channels within the wetland.
Δ Wetland condition (health) – biophysical response	
The anticipated or observed biophysical response of the wetland to the rehabilitation intervention resulting in a difference in the health of the wetland between the 'with' and 'without' rehabilitation cases. The rehabilitation outcome achieved in terms of health is determined by comparing the hydrological, geomorphological and vegetation integrity for the 'with rehabilitation' and the 'without rehabilitation' scenarios (Cowden and Kotze, 2008).	This reflects a change in the biophysical characteristics or 'properties' of the wetland (including its structures, components, processes, e.g., vegetation composition, movement of both surface and sub-surface water, nutrient cycles, biological diversity), which form the basis of, or maintain, potential services utilisable by humans, but does not yet consider potential or actual human users or benefits (Bastian et al., 2012). Example[112]: Difference in hydrological (e.g., pattern of water flows) and vegetation (e.g., vegetation cover) integrity between the 'with' and 'without' rehabilitation cases affecting the uptake of nutrients by plants and the storage of nutrients (phosphorus) in soil organic matter.

112 The nutrient retention service example is continued throughout the table to illustrate the elements of the value pathway and draws from the Mthinzima wetland rehabilitation case study.

Definition	Indicators and examples
Δ Wetland service potentials - Δ Capacity to supply	
The difference in the capacity of the wetland to provide ecosystem services between the 'with' and 'without' rehabilitation cases. Wetland service potentials are distinguished from wetland services to differentiate between the *capacity* of a wetland to supply a service and the level or quality *actually utilized or desired* by people (following Bastian et al., 2012).	'Potentials' reflect the opportunity for wetland structure and processes to contribute to human well-being / meet human needs. The 'potential' only becomes a 'service' in the presence of demand (which may be derived from a demand for the related benefit). Example: Increased effectiveness of the wetland for nutrient retention as a direct result of alterations to the hydrological regime and vegetation cover achieved through the rehabilitation interventions (Δ in capacity of the wetland to provide a nutrient retention service).
Δ Wetland services - Demand	
The additional level of service, for which there is a demand, between the 'with' and 'without' rehabilitation cases. Wetland services are the actually utilized or desired (actively or passively) outputs of wetland structure and process that contribute to human well-being (after Fisher et al., 2009; Bastian et al., 2012). Key being that wetland services, while being ecological phenomena, are "contingent on particular human activities or wants" (Boyd and Banzaf, 2007:621). For there to be a service, there must be a human beneficiary, a demand for the ecological phenomena (Wainger and Manzotta, 2011). In the context of rehabilitation, this interpretation is taken further to mean that, while the capacity of a wetland to provide a service (degree or quantity) may be greater with rehabilitation, only that degree or quantity of the 'potential' for which there is a demand is 'realized'. Where the difference in service capacity between the with and without rehabilitation cases is fully demanded then the wetland service potentials and wetland services will be equal.	Characteristics of wetland services particularly relevant to valuation[113]: • Wetland systems are multifunctional and may provide a range of services; • A single service may be linked to multiple benefits; • A benefit may derive from a combination of services; • Some services or benefits may be complementary, while others may be rival or mutually exclusive (e.g., 'use' of one service may compromise 'use' or benefit of another); • A wetland service is not necessarily beneficial across all individuals and contexts; • Wetland services and benefits are not always co-located in space and time – a service generated in one location at one time may lead to a benefit that is realized in another location at another time; • It may be challenging, expensive or not (yet) possible to measure, monitor or project directly a change in some services (some services are measured by proxy). These characteristics require attention in valuation to avoid double-counting, while also recognizing, and accounting for, multiple services, benefits and possible dis-benefits.

[113] Drawing from de Groot et al. (2002); Boyd and Banzhaf (2007) and Fisher et al. (2009).

Definition	Indicators and examples
	Example: The additional nutrient retention capacity 'utilised' to achieve a downstream water quality objective / target. Nutrient retention over-and-above that required to achieve the target may not be regarded as a service (but, consider future use/demand and option value).
Benefit outcomes (specific to the ecosystem service Δ) – Access (and demand)	
The (positive) tangible or intangible human well-being[114] outcomes/contribution directly related to the difference in wetland condition and service provision between the 'with' and 'without' rehabilitation cases. Benefits stem from ecosystem services in combination with additional inputs and conditions that enable people to derive benefit from the service (Ribot and Peluso, 2003; Wainger and Manzotta, 2011; Turner et al., 2016). Well-being outcomes of changes in wetland condition and / or service provision are not unambiguously beneficial to all groups of people; certain outcomes may be perceived as positive to some groups and negative to others. There may be dis-benefits associated with wetland ecosystems (e.g., pests, flooding) which may be generated or increased through rehabilitation	Enhancing ecosystem service supply does not necessarily translate directly into a benefit. To identify how, and for who, changes in wetland health and processes generate benefits, both supply, demand and access conditions must be interrogated. 'Factors of access', such as inputs in the form of other 'capitals', legal rights, socio-cultural norms, demographic characteristics, mediate whether benefits can be realized from ecosystem service potentials and the distribution of benefits across different social groups / individuals[115]. Example: Improved quality of water for human use, specifically a reduction in the load of nutrients to a water supply impoundment and the associated reduction in eutrophication risk. Factors of access, in this example, influencing the ability of people to derive a benefit from the improved nutrient retention of a wetland, include the additional inputs of built infrastructure and human capital of the formal water supply system and access to water from the system in terms of people's location (i.e., within the water supply area) and physical access to a piped water source.
Values associated with the Δ in condition – value diversity	
Values specific to the difference in the biophysical properties and ecosystem services of the wetland, and the associated benefits thereof, as a direct result of the rehabilitation (in contrast to the 'value' of the wetland).	Examples Economic value: the value of a reduced nitrogen load to a water supply impoundment (benefit outcome) associated with increased nitrogen retention of a wetland (ecosystem service, fully demanded) as a result of the rehabilitation, measured by proxy as the cost of equivalent nitrogen abatement achieved through an alternative means (replacement cost method).

[114] Well-being is commonly conceptualized as consisting of five inter-related constituents: security, basic material for a good life, health, good social relations and freedom of action and choice (MA, 2005a). Several authors have, however, indicated a need to better understand the concept of well-being and good 'quality of life' and how it is achieved, particularly in the context of ecosystem services and 'natures contributions to people (e.g., Fish, 2011; Díaz et al., 2015).
[115] The ability of people to derive benefit from wetland services is referred to as 'access', largely following the definition by Ribot and Peluso (2003), but further emphasizing the need for additional inputs (e.g., human and built capital) in order to realize benefits (see for example Turner et al., 2016).

Definition	Indicators and examples
'Values' are interpreted as the multiple, and not necessarily commensurable[116], ways in which wetland properties, services and associated benefits are important to individuals and social groups. This 'value diversity' emerges from the varied and dynamic interactions between people and wetlands. Values are expressed, formed or measured through valuation processes and methods. Articulated values are the expression of particular notions of the values (value domains/dimensions/types) people attribute to wetlands through valuation processes (Arias-Arévalo et al., 2018). Articulated values are shaped by the method of valuation and the context within which they are articulated (Vatn, 2005; O'Neill et al., 2008; Spangenberg and Settele, 2016) and are scale (spatial and temporal) and stakeholder specific. Values may be expressed or measured in narrative, numerical, non-monetary and monetary forms.	Ecological value: the additional contribution to biodiversity maintenance through the enhanced provision of wetland habitat as a result of the rehabilitation, measured as a gain in area of functional wetland habitat derived from a rating of the ecological integrity of the wetland 'with' and 'without' rehabilitation (following Cowden and Kotze, 2008). Broadly, the ecological value of rehabilitation is related to the contribution it makes to the recovery of a degraded system towards an intact condition in terms of its ecological attributes of integrity and resilience (Clewell and Aronson, 2007). Relational value: the contribution of the rehabilitation in strengthening the sense of purpose (meaningful occupation) and identity tied to wetland farming and supporting the continuity of traditional livelihoods and culture, assessed through interview / survey / focus group discussion methods with both wetland and non-wetland farmers[117].
Value 'pathways'	
Conceptual linkages between the outcomes of wetland rehabilitation and the 'values' of these outcomes. Value pathway (a) adopts the ecosystem service framing to reflect the links between a change in the condition of the wetland as a result of rehabilitation, the resulting effect on ecosystem services and associated benefits and values. Value pathway (b) recognizes that there are different dimensions of ecosystem value that do not necessarily fit within the ecosystem services framing. These 'value pathways' are a simplification which masks the complexities, interactions and feedbacks between these 'links' and the knowledge limitations and challenges to characterizing and quantifying these relationships.	Example (a) An increase in the retention of nutrients by the wetland (service) leading to a reduction in the load of nutrients to a water supply impoundment (benefit) reducing the risk of increased water treatment costs (value) for potable water users/customers (beneficiaries). (b) Enhanced intrinsic value of the wetland reflecting the value the wetland 'holds' that is independent of whether humans have any preference for, or derive any benefit from, the wetland.
Context, scale and beneficiaries	

[116] Value incommensurability, in the context of environmental values, implies a plurality of values that cannot be compared along some common cardinal scale such as money or energy, but may be comparable along an ordinal scale (Martinez-Alier et al., 1998; Aldred, 2006) (section 2.1.3.4).

[117] These relational value outcomes were evident in the Manalana wetland rehabilitation case study where focus group participants explained that wetland farming is 'part of life'; it is part of their identity and the social fabric of the community.

Definition	Indicators and examples
Context refers to the biophysical, societal and management circumstances[118] that form the setting of the rehabilitation and influence the human-wetland-rehabilitation interactions. These circumstances determine the actual rehabilitation benefits that can be realized (Wainger et al., 2001) and how the rehabilitation is perceived and valued and by who (O'Neill et al., 2008; Spangenberg and Settele, 2016). Human-wetland interactions occur at different spatial and institutional scales (Turner et al., 2000; Hein et al., 2006), across time (Barbier et al., 1997; Fisher et al., 2009), and are conditioned by power relations (Campbell and Olsen, 1991; Linton and Budds, 2014). Accordingly, rehabilitation benefits and values are specific to the landscape and stakeholder / beneficiary context (subjective, contingent on culture and history), vary across scales, and are dynamic across time.	The benefits, likely beneficiaries, and the associated vales of wetland rehabilitation are conditioned by a range of social contextual factors at the catchment scale and beyond. Effectively, these contextual factors, such as the current and historical use of the wetland and of the land adjacent, upstream and downstream of the wetland and the proximity of the wetland to human settlements and the characteristics of the settlement are an indication of the potential use and value of wetland services, along with 'factors of access' (with which they overlap)[119]. Example: The construction of a waste water treatment works upstream of a wetland could influence the value of an increase in the capacity of the wetland to provide water quality enhancement services as a result of the rehabilitation. In this case, the 'technical' context of future infrastructure development is particularly relevant in considering the value of the rehabilitation.
Valuation process	
The process of characterizing, representing or measuring the value(s) of (importance attached to) the difference in wetland condition 'with' and 'without' the rehabilitation. A broken line is used in the framework diagram to emphasise that valuation assessments will likely only reflect a sub-set of potential values associated with the rehabilitation outcomes.	The process involves conceptualizing the 'value pathway' to identify biophysical changes, benefit outcomes, value types and 'value to who'; prioritizing benefit outcomes and value types for assessment / measurement; selecting appropriate valuation approaches and methods, and interpreting value estimates. These 'aspects' are interlinked, which may require an iterative approach as more information is generated.

[118] Further the 'landscape context', the values individuals hold or assign are influenced and mediated by their material circumstances, worldview and culture, history and the knowledge and understanding they have at the time (O'Neill et al., 2008; Chan et al., 2016; Mensah et al., 2017; Arias-Arévalo et al., 2018).
[119] Several contextual factors particularly relevant to wetland rehabilitation valuation are presented in Appendix 7.2.

Appendix 7.2: Contextual aspects relevant to wetland rehabilitation valuation

During the course of this research, a number of contextual aspects relevant to wetland rehabilitation valuation were identified. Following the STEEP context framework, the rehabilitation aims and design and the evaluation purpose and management / decision-making information are key aspects of the 'technical' context and important in identifying expected rehabilitation outcomes and prioritizing benefits and values for assessment. The ecological context includes site-specific factors, such as wetland type, catchment scale wetland extent and connectivity and the level and drivers of degradation.

Social contextual factors identified through the research include, the current and historical use of the wetland and of the land adjacent, upstream and downstream of the wetland; the land tenure system, governance arrangements and social structures in place and prevalent power relations (formal and informal); the proximity of the wetland to human settlements and the characteristics of the settlements; water resource systems downstream of the wetland and known or historical problems with the water environment such as water pollution and stream flow disturbances; the location, or designation, of the wetland in terms of conservation schemes, protected areas, important biodiversity areas or areas of high-water yield; the extent and condition of wetlands in the region and concerns or objectives regarding wetland habitat at the regional scale and beyond (e.g., wetland rehabilitation targets of the region). Consideration of the material circumstances, culture and history, and the knowledge and understanding of individuals can provide an indication of the personal values of those affected by the rehabilitation.

Vulnerability is a further, cross-cutting, element of context. Human-orientated vulnerabilities include livelihood vulnerability (particularly relevant to consumptive uses) and vulnerability of cultural traditions (influential in considering both use and non-use values), both of which were apparent in the Manalana case study. Ecological vulnerabilities relate to, for instance, the degree to which the rehabilitation is resilient to ongoing or future pressures. Climate change vulnerability pertains to both ecological and social systems and the interactions between them.

Appendix 7.3: Factors to evaluate when considering economic valuation as an approach for articulating the importance of wetland rehabilitation

1. The 'economic good', 'private-public good' and 'market-non-market good' properties of the primary rehabilitation outcomes / benefits
Market-related goods **Direct market goods** – wetland contributions traded directly in a market (e.g., reeds sold for craft). Standard economic valuation considered reliable. Market price valuation is regarded as robust as estimates are based on observed transactions indicating actual consumer preferences. The method is limited to wetland benefits that are 'private goods' and traded directly in a market. **Inputs to market goods** - wetland attributes and functions that are inputs to the production of market goods (e.g., nursery habitat as an input to fish production). The economic value of the rehabilitation benefit is derived from a change in the production of the marketed good or service as a result of changes in wetland condition. The inputs (wetland services) can have the properties of private or public goods. Standard economic valuation considered reliable. Requires the formulation of production functions relating wetland inputs to market goods. Production function valuation is generally regarded as robust; however it can be practically challenging to define and quantify the relationship between the wetland 'input' and the market good and the influence of the rehabilitation on this relationship. **Wetland characteristics that affect consumer behaviour towards market goods** (e.g., aesthetic characteristics influencing local property prices). The rehabilitation influences these characteristics and therefore consumer behaviour. Standard economic valuation is theoretically legitimate, but reliability of economic value estimates debated. Economic valuation methods infer individual preferences for wetland benefits from observed consumer behaviour (revealed preference methods). Methods are founded on a relationship of weak complementarity between wetland characteristics and a market good. Provides information about current relationships, but not explicitly about changes in wetland conditions. Very specific context for method application, requires detailed data, often impractical.
Non-market goods **Economic goods** - wetland contributions that provide 'utility' and are 'scarce' (economic goods / contribution to economic goods), but are not connected to a market, including goods that are non-excludable (in the sense of 'public' goods) (e.g., medicinal / cultural plants). Standard economic valuation is theoretically legitimate. Reliability of economic value estimates debated. These wetland contributions may theoretically have economic value, but the means for quantifying the level of value may be limited, expensive and /or provide low confidence estimates. Economic valuation methods for these types of goods are associated with a hypothetical context and aim to elicit economic values through stated preferences (rather than observed preferences) using questionnaire-based approaches. Evolving approaches, such as combining deliberative processes with standard economic valuation methods, aim to address some of the challenges with valuing ecosystem contributions not connected to an existing market. **Non-economic goods** - wetland attributes, functions and human-wetland interactions which do not, strictly, have the properties of economic goods. This group includes forms of human-wetland interactions and value dimensions generally outside of the economic calculus, such as ecological and social resilience, equality, stewardship and other relational and moral / held values, as well as aspects perceived as 'abundant' rather than 'scarce'.

Economic valuation is limited and its reliability debateable. Stated-preference methods may be able to 'measure' a portion of these values in the sense that people may be 'willing to pay', hypothetically, towards wetland conservation or rehabilitation to maintain these other dimensions or types of 'values' or to ensure that an ecosystem and / or its contributions remain 'abundant'. However, the 'valuer' may feel that such aspects or relationships cannot be framed from an 'exchange' perspective.

Valuing aspects perceived as 'abundant' from an economic value perspective also poses challenges in that economic value is measured at the margin and therefore the 'next additional unit' of an abundant good has a relatively 'low' economic value (as is the case for water, whereas water is a fundamental constituent of life). Other forms of valuation and means of expressing the importance of wetlands, and their rehabilitation are likely to be more suitable in this context, or can be used as complements to, or in combination with, economic valuation methods to support the interpretation of value estimates or improve their reliability.

Unique goods or services with no substitutes
By definition, (neoclassical) exchange value implies some degree of substitutability. Where substitutability is uncertain, or from a strong sustainability perspective, economic valuation is not appropriate (Spangenberg and Settele, 2016).

A strong sustainability view holds that there is a qualitative difference between manufactured and natural capital, and specifically, that there are elements of natural capital that are 'critical' and their destruction cannot be compensated through investing in the other 'capitals' (Ekins et al., 2003; Neumayer, 2012). Ekins et al. (2003) suggested that in some cases substitutability between forms of capital exists; that is, the welfare derived from different 'capitals' are commensurable, but that this is not always the case.

2. Whether the rehabilitation is associated with non-critical changes in wetland state and incremental changes in wetland attributes and functions

Economic value is measured as marginal utility - the desire for *an additional unit* of the 'good' (object or service) and assumes an existing stable stock of the good and incremental changes in the flow of goods. In the case of a wetland system on the verge of an ecological regime change, economic value estimates become unreliable and economic valuation is not appropriate.

3. Whether the rehabilitation outcomes are reliably and feasibly measurable in terms relevant to human well-being (and recognized by people)

The application of economic valuation methods necessitates that there is a well-defined difference in wetland condition and function as a result of the rehabilitation and a corresponding measurable benefit to people that individuals recognize and can respond to. In practice, the economic valuation of wetland rehabilitation is limited to those aspects associated with biophysical outcomes that it is possible and feasible to quantify and which people recognize as contributing to their well-being.
Combining deliberative processes with standard economic valuation methods can address issues of a lack of familiarity with or recognition of the contribution of wetlands to people.

If the relationship between the ecological change and the human relevant outcome cannot be specified and the human benefit relevant outcomes cannot be measured, then economic valuation methods cannot be applied. The degree of uncertainty or extent of missing information associated with measurements of biophysical changes as a result of the rehabilitation influences whether economic valuation is possible and reliable. A further aspect here is whether the biophysical outcomes and affected services can be meaningfully / practically specified in terms of individual units and the addition of individual units between the with and without rehabilitation cases quantified. Economic valuation methods are not suited to valuing 'lumpy goods' (i.e., wetland attributes or functions and associated benefits that cannot be measured incrementally).