

EXPLORING THE SUITABILITY OF CAUSAL LOOP DIAGRAMS TO ASSESS THE VALUE CHAINS OF AQUATIC ECOSYSTEM SERVICES: A CASE STUDY OF THE BAVIAANSKLOOF, SOUTH AFRICA.

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

MASTER OF SCIENCE (MSc)

Of

RHODES UNIVERSITY

BY:

JONATHAN MARK RAWLINS

SUPERVISED BY:

PROF G.C.G. FRASER

DR W.J. DE LANGE

Department of Economics and Economic History

Rhodes University

Grahamstown

South Africa

April 2016

ABSTRACT

Healthy, functioning aquatic ecosystems are fundamental to the survival and development of any nation, particularly so for water-stressed countries like South Africa. Aquatic ecosystem services (AESs) are becoming increasingly recognised for their importance to society with regards to the ecological goods and services they provide in terms of health, social, cultural and economic benefits. The development of markets for AESs begins with a clear understanding of the nature and extent of the goods and services provided by aquatic ecosystems. However, an inclusive understanding of AESs and their associated values is currently lacking in South Africa.

Although flows of ecosystem services provide a nearly limitless set of valuable properties, a large proportion of their services remain unpriced or inaccurately priced through traditional neo-classical markets. This often results in market failure, as these markets do not reflect the full social costs and/or benefits of ecosystem services. This provides incentive to identify and develop a tool to bridge the gap between ecosystem service valuation and practical, sustainable management recommendations for improving the provision of ecosystem services and their associated markets.

This study explores the suitability of causal loop diagrams (CLDs) to assess the value chains of AESs in South Africa within the context of a case study. AESs do not usually have finite market values nor are they traded in formal markets, thus, a traditional approach to value chain analysis is unsuitable. A professional workshop environment was utilised to facilitate a transdisciplinary approach towards identifying relevant AESs and their complex inputs, interactions and trade-offs. Numerous CLDs were developed in an effort to map the complex relationships between these AESs and their associated inputs, which formed the basis to attempt subsequent scenario analyses and 'alternative' value chain analyses. The findings of this study show that CLDs have the potential to qualitatively identify challenges and opportunities within the value chains of AESs. Thus, the use of such 'alternative' value chain analyses can directly contribute towards the development of recommendations for improving sustainable management of aquatic ecosystems.

Keywords: aquatic ecosystem services, causal loop diagram, value chain analysis, sustainable ecosystem management.

DECLARATION

This thesis has not been submitted to a university other than Rhodes University, Grahamstown, South Africa. The work presented here is that of the author, unless otherwise stated.

Signed  Date 08/09/2016

TABLE OF CONTENTS

<i>Abstract</i>	ii
<i>Declaration</i>	iii
<i>Table of Contents</i>	iv
<i>List of Figures</i>	vii
<i>List of Tables</i>	ix
<i>List of Abbreviations</i>	x
<i>Acknowledgements</i>	xii
 CHAPTER 1: INTRODUCTION	1
1.1. Research context.....	2
1.2. Problem statement	6
1.3. Aims and objectives	6
1.3.1. Aim	6
1.3.2. Objectives	6
1.4. Structure of thesis.....	6
 CHAPTER 2: ECOSYSTEM SERVICES	8
2.1. Introduction	9
2.2. The concept of ecosystem services and definitional issues	9
2.3. Ecosystem service classification and typology	16
2.4. Ecosystem service identification.....	22
2.5. Ecosystem service assessment, management and sustainability tools.....	22
2.6. Aquatic ecosystem services	29
2.7. The concept of value and valuation of ecosystem services	32
2.7.1. Evaluating aquatic ecosystem services in south africa	35
2.7.2. Challenges and limitations of ecosystem service valuation	35
2.8. Summary	37
 CHAPTER 3: SYSTEM DYNAMICS, CAUSAL LOOP DIAGRAMS & VALUE CHAIN ANALYSES ..	38
3.1. Introduction	39
3.2 System dynamics.....	39
3.2.1. Causal loop diagrams and theory	44

3.2.2. Benefits and shortfalls of causal loop diagrams	49
3.3. Value chain analysis	52
3.3.1. Value chain analysis approaches	54
3.3.2. Issues and limitations surrounding value chain analysis	56
3.3.3. Ecosystem services and value chains	58
3.3. Summary	60
CHAPTER 4: THE BAVIAANSKLOOF CATCHMENT	61
4.1 Rationale for study area selection	62
4.2. Physical attributes of the study area	63
4.2.1 Location and topography	63
4.2.2 Climate and hydrology	64
4.2.3 Geology	66
4.2.4 Surface water and surrounding catchments	67
4.2.5 Vegetation	68
4.3 Land use	68
4.4. Summary	69
CHAPTER 5: RESEARCH METHODS AND STUDY DESIGN	70
5.1. Overview and study design rationale	71
5.2. Phase 1: causal loop diagram development	73
5.2.1. Expert workshop	73
5.2.2. Professional and site verification	80
5.3. Phase 2: causal loop diagram analysis	81
5.3.1. Scenario analyses	81
5.3.2. Value chain analyses	83
5.4. Limitations and strengths	85
5.5. Ethical considerations	87
5.6. Summary	87
CHAPTER 6: SCENARIO AND VALUE CHAIN ANALYSES	88
6.1 Rationale	89
6.2. Scenario analyses	92

6.2.1. Fire	92
6.2.2. Drought	94
6.2.3. Land-use change	96
6.2.4. Catchment restoration.....	98
6.3. Value chain analyses	100
6.3.1. Water provision	102
6.3.2. Flood attenuation	105
6.3.3. Aquatic ecosystem health.....	107
6.4. Strengths and limitations.....	109
6.5 Summary	112
CHAPTER 7: RECOMMENDATIONS AND CONCLUSIONS	113
7.1. Recommendations	114
7.1.1. Future research.....	114
7.1.2. Practical applications	117
7.2 Conclusions	118
REFERENCE LIST.....	122
APPENDIX	147
1. Causal Loop Diagram Development Contributing Participants	148
2. Aquatic Ecosystem Service Descriptions and Classifications.....	150
3. Examples of Causal Loop Diagram Development Versions	156

LIST OF FIGURES

Figure 1: Interaction between built, social, human and natural capital required to produce human well-being	12
Figure 2: The Ecosystem Service Cascade Model	13
Figure 3: The pathway from ecosystem structure and processes to human well-being.....	15
Figure 4: Biodiversity, ecosystem functioning and drivers of change	17
Figure 5: The distinction between intermediate services, final services and benefits illustrated by the stylised relationship between supporting, regulating, provisioning and cultural services	18
Figure 6: Good and service characterisation continuum from rival to non-rival and excludable to non-excludable.....	20
Figure 7: Relationship of hydrologic ecosystem processes to hydrologic services.....	31
Figure 8: Conceptual ecosystem service cascade framework for hydrological services provision by forests.....	32
Figure 9: Simplified overview of the WRC aquatic ecosystem service evaluation framework	35
Figure 10: Three viewpoints of cause and effect	41
Figure 11: Illustration of variable types and causal links in system dynamics	42
Figure 12: Causal loop diagram (left) and stock/flow diagram (right) illustrating the bathtub filling example explained in Figure 11	43
Figure 13: Hydrology subsystem causal loop diagram	46
Figure 14: Simplified causal loop diagram illustrating the impact of floods on communities in Northern Ghana	47
Figure 15: Causal loop diagram illustrating water security in the Sand River Catchment	48
Figure 16: Porter's classical value chain of the firm	55
Figure 17a: Location of the Baviaanskloof Catchment.....	64
Figure 17b: Digital Elevation Model of the Baviaanskloof Catchment.....	64
Figure 17c: Map of the Baviaanskloof Quaternary Catchments Showing the Prevailing MAP and MAE.....	65
Figure 17d: Catchments Surrounding the Baviaanskloof	67
Figure 18: Study design flow chart	71

Figure 19: Causal Loop Diagram Illustrating Select Aquatic Ecosystem Services, Processes and Functions in the Baviaanskloof Catchment	90
Figure 20a: Fire Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment	93
Figure 20b: Drought Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment	95
Figure 20c: Land-use Change Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment	97
Figure 20d: Catchment Restoration Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment	99
Figure 21: Four Tier Traditional Value Chain Example for Water Provision	104
Figure 22a: Examples of Linear Causal Pathways Affecting Water Provision	104
Figure 22b: Examples of Linear Causal Pathways Affecting Flood Attenuation	106
Figure 22c: Examples of Linear Causal Pathways Affecting Aquatic Ecosystem Health	108

APPENDIX

Figure 1: First aquatic ecosystem service causal loop diagram version	156
Figure 2: Sixth aquatic ecosystem service causal loop diagram version	157
Figure 3: Thirteenth aquatic ecosystem service causal loop diagram version	158

LIST OF TABLES

Table 1: Ecosystem Service Categories in MA, TEEB and CICES	19
Table 2: Advantages and limitations of causal loop diagrams	13
Table 3: Process steps that make up phase 1 and 2 of the study design.....	72

APPENDIX

Table 1: Expert workshop participants.....	148
Table 2: Professional verification meeting participants.....	149
Table 3: Final aquatic ecosystem services descriptions, classifications and beneficiary groups	150
Table 4: Intermediate aquatic ecosystem service descriptions, units of measurement and causal linkages	151
Table 5: Anthropogenic variable descriptions, units of measurement and causal linkages.	155

LIST OF ABBREVIATIONS

AES	Aquatic Ecosystem Service
ARIES	Artificial Intelligence for Ecosystem Services
BMR	Baviaanskloof Mega Reserve
BBN	Bayesian Belief Network
CFR	Cape Floristic Region
CLD	Causal Loop Diagram
CICES	Common International Classification of Ecosystem Services
DEM	Digital Elevation Model
ES	Ecosystem Service
ESB	Ecosystem Services Benchmark
ESVCA	Ecosystem Service Value Chain Analysis
GIS	Geographic Information System
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
MAE	Mean Annual Evaporation
MAP	Mean Annual Precipitation
MMODS	Mental Model Of a Dynamic System
masl	meters above sea level
MA	Millennium Ecosystem Assessment
MIMES	Multiscale Integrated Model of Ecosystem Services
NWA	The South African National Water Act

NWRS	National Water Resource Strategy
PES	Payment for Ecosystem Services
SWAT	Soil and Water Assessment Tool
SFD	Stock/Flow Diagram
SWSA	Strategic Water Source Area
STRP	Subtropical Thicket Restoration Programme
SEEA	System of Environmental-Economic Accounting
TMG	Table Mountain Group
TEEB	The Economics of Ecosystems and Biodiversity
VIC	Variable Infiltration Capacity
VSTEPP	Values Social Technology Economics Environmental Political
WRC	Water Research Commission
WFW	Working for Water

ACKNOWLEDGEMENTS

I would like to formally and graciously acknowledge the help of the numerous people and institutions that assisted me in completing this challenging research project and truly believe in the potential that this research offers. Firstly, I would like to thank Rhodes University, the Department of Economics and Economic History staff, more specifically, the Environmental and Natural Resource Economic Focus Area (ENREFA) group for the continued academic and personal support throughout the duration of this project.

To my supervisors, Prof Gavin Fraser and Dr Willem De Lange, I would like to extend my utmost gratitude and appreciation for all the time and effort you have contributed to this project. From selecting me as the appropriate student for the project and bursary, the weekly meetings and traveling all around the country to dealing with my constant emails and questions. Thank you for all of the support, feedback, criticism, motivation and patience throughout this endeavour. I have learnt an incredible amount and grown as an individual, all of which would not have been possible without your efforts.

To all the people who gratuitously gave up their time to contribute to the generation and development of the necessary diagrams and information for this research: Prof Tally Palmer, Prof Fred Ellery, Dr Tony Palmer, Rebecca Powell, Julia Glenday, Prof Jen Snowball, Dr Jane Turpie, Dr Patrick O'Farrell, Dr David Le Maitre, Dieter Van Den Broeck, Stephie Mendlesohn and Pieter Kruger. The contribution of your expertise and knowledge was invaluable to the success of this study. Furthermore, I would like to thank all of you for your willingness to assist throughout both phases of the study and buying into the research process.

I would like to formally acknowledge the South African Water Research Commission for providing the funding for this research and general guidance from the K5/2341 project steering committee: Mr B Madikizela, Mr N Adam, Dr MA Amis, Mr Y Atwaru, Dr CA Brown, Dr P Chivenge, Mr J Crafford, Ms T Cumming, Mr J Dini, Dr TE Kleynhans, Ms T Ndukwana, Mr TR Netshiluvhi, Dr M Mander and particularly Prof Geoff Antrobus. Furthermore, I would like to acknowledge the contribution of the CSIR in terms of facilitating the project and providing me with access to the necessary people, information and tools to conduct effective research.

I would like to acknowledge the Ecosystem Services Partnership (ESP) and the South African Young Water Professionals (YWP) for accepting this work to be presented at two competitive

international conferences which provided vital information and feedback for the development of the method. Additionally, these opportunities provided significant exposure of this research to relevant international communities which sparked further interest and contributions.

Finally, I would like to thank all of my friends and family who have always been there for me and supported me through my trials and tribulations. Your moral support has not gone unnoticed, and for this, I am forever grateful.

CHAPTER 1

INTRODUCTION



Photo: J.M. Rawlins

1.1. Research Context

Healthy, functioning ecosystems and adequate water supplies are fundamental to the survival and development of any nation, particularly so for water-stressed countries like South Africa (Brauman *et al.*, 2007; DWS, 2014). The 2030 Water Resources Group estimates that by 2030, the global population will be in excess of eight billion people and worldwide demand for water will increase by more than 40% (WRG, 2009). Freshwater is arguably the most important natural resource in the world and as essential to industry, agriculture and the environment as it is to basic human survival. It is evident that future water shortages pose one of the most significant threats to the economic, social and environmental systems that sustain modern day South Africa.

Research focusing on ecosystem services (ESs), “the benefits people obtain from ecosystems” (MA, 2005: v), has increased dramatically over the past two decades (Carpenter *et al.*, 2009; Costanza, 2008; De Groot *et al.*, 2010b; Fisher *et al.*, 2009). The ES concept is being increasingly utilised to explain the complex linkages between biodiversity, ecosystems and human welfare, thus, has been repeatedly acknowledged as a suitable model for sustainable resource management decision-making (Cilliers *et al.*, 2012; Fisher *et al.*, 2009; Vihervaara *et al.*, 2010). Furthermore, ESs derived from aquatic systems or aquatic ecosystem services (AESs) are becoming increasingly recognised for their importance to society with regards to the ecological goods and services they provide in terms of health, social, cultural and economic benefits (Brauman *et al.*, 2007; Ginsburg *et al.*, 2010).

Economic development and the natural environment are intrinsically and inextricably linked as natural systems provide the resources that fuel growth and ESs, which underpin both life and livelihood (Najam *et al.*, 2007; WRI, 2000). The most economically and socially vulnerable communities in South Africa are significantly dependent upon the provision of AESs such as water provision. Consequently, degradation of these ecosystems will diminish their capacity to provide these services and ultimately threaten the livelihoods of such communities (Limburg, 2009; Palmer *et al.*, 2004). Rockström *et al.* (2009) describe how humanity has transgressed at least three of nine ‘planetary boundaries’ which define the safe environmental operating conditions for planet Earth. By transgressing these boundaries, the planet is put at risk of abrupt and catastrophic environmental change (O’Neill *et al.*, 2010;

Ostrom, 2009; Rockström *et al.*, 2009). Modern day societies and economies are now under threat from the continued erosion of the stock of ESs (De Groot *et al.*, 2010a; O'Neill *et al.*, 2010; Vihervaara *et al.*, 2010; WRI, 2000).

The South African National Water Act (NWA) (Act 36 of 1998) has been internationally acclaimed as one of the most progressive and transformative pieces of water legislation in the world. The Act includes an environmental flow allocation or ecological reserve, which is the minimum flow requirements for maintaining freshwater ecosystems and basic human needs (DWA, 2013a; Palmer *et al.*, 2004). The inclusion of the ecological reserve in policy is closely linked with the concept of AESs by acknowledging the connections between environmental health and ES provision. However, various institutions related to the implementation of the NWA have failed to ensure that all components of the water value chain function efficiently and effectively and thus, environmental flow requirements have often not been adhered to (DWA, 2013a; DWA, 2013b; Schreiner, 2013).

In the face of rapidly expanding populations, increasing demand for resources, climate change, water pollution and ecosystem degradation coupled with a substantial focus on economic growth and development, it is vital to develop tools and methods to ensure long term, sustainable AES provision that is aligned with national needs and imperatives (DWA, 2013a; Palmer *et al.*, 2004). Water is a scarce resource with no substitute; thus, it necessitates careful management to ensure adequate provision of basic AESs and equitable allocation, while simultaneously addressing the needs of inclusive economic growth without threatening the integrity of natural aquatic ecosystems (Scemama and Levrel, 2015). The significance of systems thinking and trans-disciplinary research is becoming increasingly recognised for developing countries facing multiple drivers increasing the vulnerability of rural communities to ecosystem degradation. Practical outcome- and policy-oriented AES research is of the utmost importance if South Africa is to achieve the National Water Resource Strategy's (NWRS) goal of water that "is effectively and efficiently managed for equitable and sustainable growth and development" (DWA, 2013a: 12).

Although flows of ESs provide a nearly limitless set of valuable properties, a large proportion of their services remain unpriced through traditional markets (Hanley *et al.*, 2007). The inclusion of beneficiaries makes the concept of value intrinsic to all ESs, regardless of whether

or not the values are monetised (Brauman *et al.*, 2007; Fisher *et al.*, 2008). Various methods for placing monetary values on particular ESs have traditionally been used as a point of departure to facilitate ES management, as well as their integration into formal markets. Economic and environmental systems share many attributes; however, the valuation of ESs has traditionally been driven by short-term human preferences (Garrod and Willis, 1999; Limburg *et al.*, 2002). Moreover, valuations of ESs face a myriad of methodological shortcomings (Barbier, 2011; Parks and Gowdy, 2013). Besides, it is often not possible to identify explicit, marketable value attributes for a particular ES (Limburg *et al.*, 2009; Pascual *et al.*, 2010; Turpie *et al.*, 2010). Thus, ESs are left without a formal market value, albeit not without value (Alpizar *et al.*, 2007; Ferraro, 2000).

Goods and services (whether derived from ecosystems or not) without a recognised market value are difficult to manage as allocations are generally made based on monetary values. This often results in market failure, as the markets for ESs do not reflect the full social costs and/or benefits of the goods and services provided (Ginsburg *et al.*, 2010; Tietenberg and Lewis, 2010). Carpenter *et al.* (2009), Ostrom (2009) and Walker *et al.* (2002) stress the need for integration between science and policy to improve the capacity to develop, implement and monitor policies that predict consequences and assess outcomes of shocks to socio-ecological systems. A higher level of AES provision is directly correlated with greater system resilience and increased human well-being (Limburg, 2009). This provides incentive to develop and/or identify a tool, which effectively crosses disciplinary divides and can be used to bridge the gap between ES valuation and practical management and policy outcomes aimed at improving sustainable provision of AESs and their associated markets.

The development of formal markets, sustainable management models and decision-support tools for AESs begins with a clear understanding of the nature and extent of the goods and services provided by aquatic ecosystems. However, an inclusive understanding of AESs, their complex interactions and associated values is currently lacking in South Africa. Understanding intricate socio-ecological systems requires a clear definition of key system components and their associated cause and effect relationships as well as a description of the relationship of the system to other systems (De Groot *et al.*, 2010b; Ford, 1999; Limburg *et al.*, 2002). Complexity is a characteristic common to all coupled human-environment systems (Loeche,

2004), the challenge to communicate the functionality of such systems lies within explaining the relationships between key elements of the system in a simple and transparent way.

Causal loop diagrams (CLDs) could potentially address some of these challenges as they are utilised to portray complex feedbacks at work within a system (Ford, 1999; Sterman, 2000). Thus, the development of an AES based CLD could facilitate graphical conceptualisation of the ESs at work within a particular system and provide a detailed understanding of the linkages between them. The flow of ESs from their source to the beneficiary would be highlighted through the linkages in the CLD, an analysis of this process would be akin to a value chain analysis. Kaplinsky and Morris (2000) describe a value chain analysis as a process that traces the value added in each step of the life cycle of a particular good or service. However, the traditional approach towards value chain analyses are geared towards linear processes and private goods. Thus, this approach would be unsuitable for analysing the value chains of AESs as they are considered public goods and have no formal, recognised market values.

CLDs could be used to model and illustrate the different feedback mechanisms and interconnected processes of select socio-ecological system components that contribute towards the provision of final AESs. This would allow for the extension of traditional value chains (for marketed and non-marketed goods and services) to include relevant environmental processes and functions. Developing an ES based model designed to assist with sustainable ecosystem management has a multitude of potential benefits. A complex model of this nature could be used to assist proactive management decision-making with regards to mitigation and/or adaption to particular system scenarios, efficient economic allocations surrounding ES management and further integration of ESs into formal markets. Increased system understanding would ideally incentivise further public and private investment into the sustainable management of ecosystems in the long-term. Ultimately, contributing towards a greater and more inclusive understanding of AESs and their complex interactions, while simultaneously bypassing the limitations and challenges of ES valuation. This is aligned with the aims and objectives of the South African National Development Plan and the three pillars of sustainable development (Feris, 2010; NPC, 2012).

1.2. Problem Statement

ESs and environmental processes are often excluded from modern day value chains analyses. Thus, there is little financial or commercial incentive to manage these resources sustainably despite their crucial contributions towards the generation of the final good or service. Currently few tools or models exist that can facilitate a value chain analysis for such complex ESs due to their public nature, limited integration into formal markets and absence of recognised market values.

1.3. Aims and Objectives

1.3.1. Aim

To explore the suitability of CLDs as a tool to assess the value chains of AESs in South Africa within the context of a case study, the Baviaanskloof catchment.

1.3.2. Objectives

Similar to the Ecosystem Services Framework (MA, 2005), this research assesses an alternative way to present and evaluate trade-offs among different scenarios of ES provision through the use of causal relationships (Brauman *et al.*, 2007). Ultimately, the aim is to determine the viability of using CLDs as an effective method to conduct value chain analyses for AESs. To effectively address the aim of this project, four specific objectives will need to be achieved:

1. Develop an AES based CLD
2. Illustrate the forward linkages and ripple effects in the value chains of final AESs
3. Identify challenges and opportunities in the value chains of final AESs
4. Recommend methods to improve AES provision and associated value chains

1.4. Structure of Thesis

Chapters 2 and 3 make up the bulk of the literature review conducted during this study and emphasise the most relevant and current theory related to the research question at hand. Chapter 2 presents an in depth literature review focused on the concept of ESs, their disputed

definitions, identification, classification and management. The notion of value is discussed in terms of the valuation of ESs. Chapter 3 provides a detailed account of relevant and current literature surrounding the field of system dynamics with a specific focus on CLDs. The processes involved in traditional value chain analyses are explained as well as the associated limitations and strengths and the potential application to ESs.

Chapter 4 presents the study area that physically contextualises the research by providing a rationale for selection of the Baviaanskloof catchment as the study area. The relevant physical attributes of the study area are discussed in relation to AESs. Chapter 5 presents and explains the reasons behind the research design and chosen methodology. The combination of approaches adopted during this research are systematically investigated and clarified for both phases of the study design.

Chapter 6 presents the general findings as well as a detailed discussion focusing on the aim and objectives of the research. The discussion is intended to illustrate the potential use of CLDs as a tool for value chain analysis and associated benefits of this. Lastly, Chapter 7 discusses numerous recommendations for future research, how the model could be improved and/or altered for various uses/applications and where the model could fit into existing management and decision-making models/bodies. An overall synopsis of the thesis is provided which emphasises the key outcomes that have emerged from the research and final conclusions are made.

CHAPTER 2

ECOSYSTEM SERVICES



Photo: J.M. Rawlins

2.1. Introduction

The concept of and theory around ESs has developed into an important model linking human welfare to the functioning of natural ecosystems (Fisher *et al.*, 2009; Müller and Burkhard, 2012; TEEB, 2010). Relevant and accurate policy and management decision-making is largely reliant upon the understanding of these linkages (Costanza *et al.*, 1997; MA, 2005; TEEB, 2010). So too is the development of a tool to analyse these linkages (value chains) and promote effective and sustainable ES management. This chapter discusses the notional and definitional issues around ESs, recognised ES classification and identification procedures and renowned ES assessment and sustainable management tools. AESs are explained and defined in terms of the research aim and limitations of ES valuation methods are briefly discussed.

2.2. The Concept of Ecosystem Services and Definitional Issues

Despite numerous attempts to develop a standardised classification scheme for ESs, there remains no consistent, agreed upon definition for the term (Boyd and Banzhaf, 2007; Fisher *et al.*, 2009). Research focusing on ESs has increased dramatically in recent time, resulting in a myriad of proposed ES definitions for various applications (De Groot *et al.*, 2010b; Haines-Young and Potschin, 2014; Vihervaara *et al.*, 2010). However, there is little information available in the literature distinguishing how ESs should be defined (Barbier, 2007; Boyd, 2007; Fisher *et al.*, 2009; Wallace, 2008). Some of the most widely used and cited definitions include:

- “the benefits human populations derive, directly or indirectly, from ecosystem functions” (Costanza *et al.*, 1997: 253)
- “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” (Daily, 1997: 3)
- “benefits people obtain from ecosystems” (MA, 2005: v)
- “the direct and indirect contributions of ecosystems to human well-being” (TEEB, 2010: 33)
- “the contributions of ecosystems to benefits used in economic and other human activity” (EC, 2013: 164)

Although these definitions are similar in many ways, there are subtle differences that require robust explanations regarding the reasoning behind the choice of definition. The definitions

differ in terms of the specificity of various terms such as ‘benefits’, ‘contributions’, ‘goods’, ‘services’, ‘direct’, ‘indirect’, ‘function’, etc. The semantic differentiation between the terminology is so nuanced there are even debates regarding the differences between ecosystem ‘function’ (viewed anthropocentrically to be goal oriented) and ecosystem ‘functioning’ (viewed ecocentrically without specific goals) (De Groot *et al.*, 2002; Fisher *et al.*, 2009; Jax, 2005).

Using the ES model requires defining an ‘ecosystem’ and distinguishing between these contested terms with specific reference to the application at hand (Carpenter *et al.*, 2009; COWI, 2014; Costanza *et al.*, 2014; Fisher *et al.*, 2009; Nunes *et al.*, 2014; ONEMA, 2011; TEEB, 2010; Wallace, 2008). The Convention on Biological Diversity (UN, 1992: 3) defines an ecosystem as “a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit”, this is the commonly accepted definition. The abovementioned EC (2013) definition of ESs has gained increased support in recent time due to the awareness of the need for such distinctions in terminology (COWI, 2014; Fisher *et al.*, 2009). ONEMA (2011) make a minor distinction between ESs and ‘ecological services’, the latter is preferred as it can be applied to various composite spatial units which may group together a number of different ecosystems. This distinction is based on the fact that ecosystems can be defined at a range of spatial scales (e.g. pond vs tundra), are heavily interconnected and are impacted by processes that function over variable spatial and temporal scales (Braat *et al.*, 2012; EC, 2013).

Boyd and Banzhaf (2007: 619) provide an alternative definition of final ESs aimed at advancing performance systems and environmental or ‘green’ accounting: “components of nature, directly enjoyed, consumed or used to yield human well-being”. This definition simplifies many of the terms within the definition and provides a more specific and streamlined alternative to the aforementioned ‘broad’ definitions. This definition highlights how ESs are ecological components consumed and/or used directly to yield human well-being as opposed to simply being the benefits humans receive from ecosystems (Boyd and Banzhaf, 2007; De Groot *et al.*, 2002; Fisher *et al.*, 2009; Sagoff, 2011; Van Wilgen *et al.*, 2008). This definition notably excludes indirect process, functions and services, elucidating the differences between the concepts of ‘services’ and ‘benefits’ (Fisher *et al.*, 2009).

In an attempt to extend the definition proposed by Boyd and Banzhaf (2007), Fisher *et al.* (2009: 645) proposed that ESs are “the aspects of ecosystems utilized (actively or passively) to produce human well-being”. It is important to note that this definition requires that ESs are ecological phenomena of some kind that can be directly or indirectly utilized (Fisher *et al.*, 2009). Whereas Boyd and Banzaf (2007) view ESs as directly consumable end points, this definition includes ecosystem ‘structure’ and ‘functions’ whether utilised directly or indirectly.

Johnston and Russell (2011) highlight how most ES analyses fail to effectively distinguish between final ESs and ecological functions and processes that provide benefits indirectly, referred to as intermediate ESs. Two primary sources of ambiguity arise in the classification of values associated with final vs intermediate ESs. Firstly, the lack of widely applicable and simple guidelines to aid in deriving a replicable and consistent ES classification, and secondly, the application of universal final ES typologies that apply to all beneficiaries (Johnston and Russell, 2011).

The concept of humans as beneficiaries to these ‘environmental services’ means that these natural ‘functions’ and/or ‘processes’ are considered to be ESs (Fisher *et al.*, 2009; Limburg *et al.*, 2002; ONEMA, 2011; Perman *et al.*, 1996). However, ecosystem ‘structure’, ‘function’, and ‘services’ are not identical. Most ecosystem ‘structures’ and ‘processes’ do provide ESs but they are not necessarily the same thing. For example, Fisher *et al.* (2009), Johnston and Russell (2011) and Limburg (2009) describe how nutrient cycling is an example of a service that is used indirectly by people as one of the outcomes is clean water, a service which people use directly. Thus, clean drinking water can be considered to be a final ES or direct ‘benefit’ derived from different intermediate ESs. The importance of differentiating between final and intermediate ESs is becoming increasingly recognised in ES science as it is important to avoid double counting of services among other ambiguities (COWI, 2014; EC, 2013; Johnston and Russell, 2011; ONEMA, 2011).

Figure 1 illustrates the interactions required to produce human well-being, which is an integral factor in defining and understanding ESs. Built and human capital (the economy) are embedded within social capital (society), which is embedded within natural capital (nature) (Costanza *et al.*, 2014). The economy is a sub-system of the environment because the natural

environment provides all inputs into the economy and all by-products and waste are returned to the environment (O'Neill *et al.*, 2010). This diagram demonstrates how ESs are the relative contribution of natural capital towards human well-being, despite not flowing directly. Thus, when addressing ESs, it is imperative to use a transdisciplinary approach as there are a multitude of factors which determine how ESs contribute towards human well-being throughout social-ecological systems (Carpenter *et al.*, 2009; Costanza *et al.*, 2014; COWI, 2014; EC, 2013; MA, 2005; Van Wilgen *et al.*, 2008; Wallace, 2008).

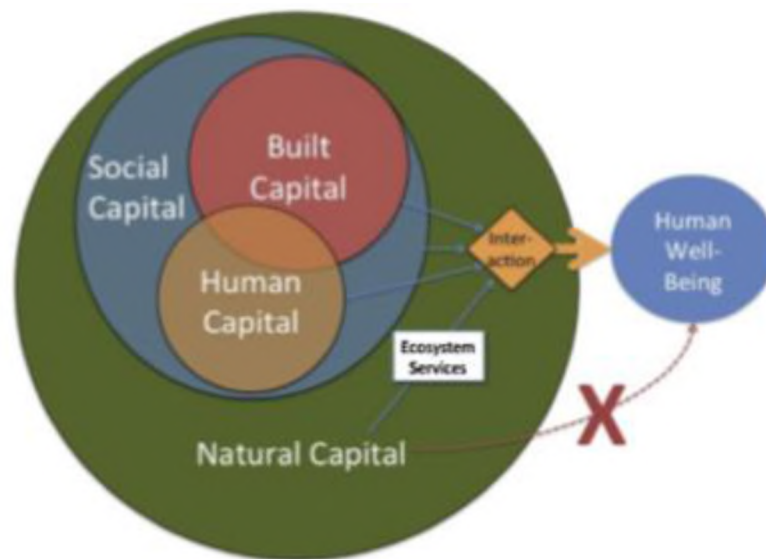


Figure 1: Interaction between built, social, human and natural capital required to produce human well-being (Costanza *et al.*, 2014).

The ES cascade (Figure 2) is a transdisciplinary, iterative and analytical model used to systematically summarise much of the logic that underpins the modern ES paradigm (COWI, 2014; Potschin and Haines-Young, 2011). The purpose of the ES cascade is to highlight the necessary elements that need to be considered in any ES analysis, as well as the relationships between them (Potschin and Haines-Young, 2011). This model has been discussed and adapted by others (i.e. De Groot *et al.*, 2010a (Figure 3); Salles, 2011) as it presents the notion that there is some form of 'production chain' linking biophysical and ecological processes and structures on the one side and components of human well-being on the other, with a sequence of intermediate stages between (Potschin and Haines-Young, 2011). This conceptual model further illustrates the differentiation between ecosystem 'structure', 'function', 'service', 'benefit' and 'value'. The differentiation between these concepts is linked

to the ‘production boundary’, which graphically illustrates a simplification of the interface between social and economic systems and the natural environment (COWI, 2014; EC, 2013). ‘Values’ are separated from ‘benefits’ because if benefits are considered as ecosystem generated welfare gains, then these ‘benefits’ may be valued differently by different stakeholders at different spatial and temporal scales (Fisher *et al.*, 2009; Potschin and Haines-Young, 2011).

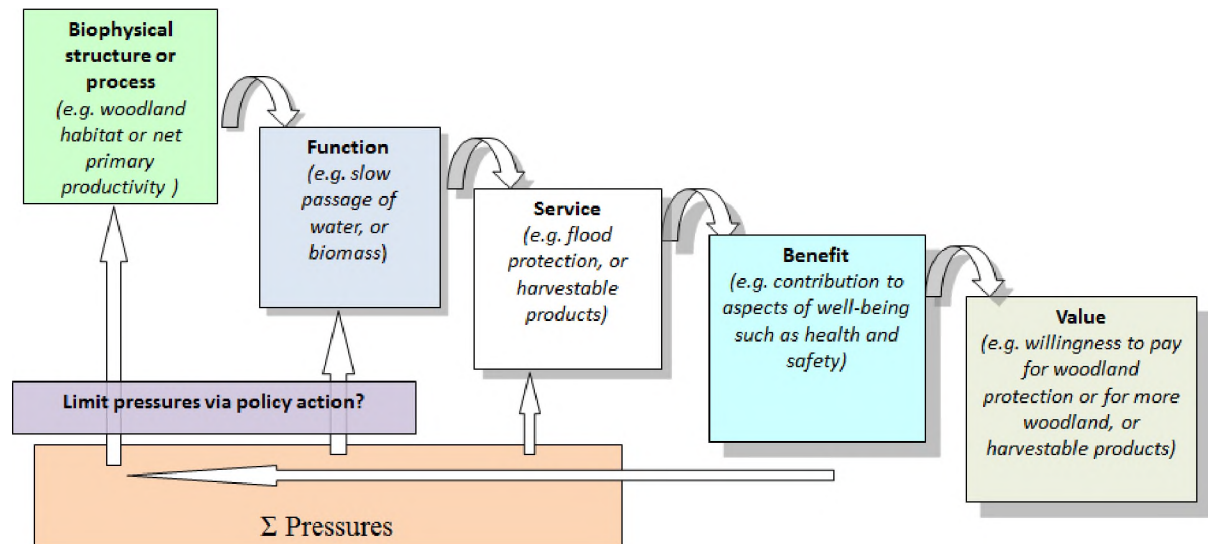


Figure 2: The Ecosystem Service Cascade Model (Haines-Young and Potschin, 2011).

Considering the ES cascade (Figure 2), the term ‘function’ is used to indicate the capacity of a particular ecosystem to provide or do something that is useful to people in some way or another (Brown *et al.*, 2007; Costanza *et al.*, 1997; Daily, 1997; De Groot *et al.*, 2002; Potschin and Haines-Young, 2011). Jax (2005; 2010) emphasises the differences in understanding of the term ‘function’ and the importance of choosing the appropriate definition in a particular context. Hence, Wallace (2007) considers ecosystem ‘functions’ and ‘processes’ as the same concept, to simplify the assessment and avoid confusion. COWI (2014), Fisher and Turner (2008) and Fisher *et al.* (2009) have further categorised the elements on the left-hand side of the ES cascade diagram (Figure 2) as ‘intermediate services’ (biophysical structure/process and function), the elements on the right as ‘goods and benefits’ (benefit and value) and the ‘service’ component as ‘final services’. Balmford *et al.* (2011) classify ESs into three categories, namely ‘ecosystem benefits’, ‘core ecosystem processes’ and ‘beneficial ecosystem processes’, these are then ranked according to their contribution towards human welfare. It is evident from the variety of approaches, understandings and categorisations, including the

ecosystem cascade, that a central challenge is understanding the linkages between ecological systems and human well-being (Potschin and Haines-Young, 2011).

The fundamental principle of the ES paradigm remains; a human beneficiary needs to be identified for any ES to exist. Thus, it is essential to distinguish between final ESs that contribute to human well-being and the ‘intermediate ecosystem structures and functions’ (intermediate ESs) that give rise to them (Figure 2) (Balmford *et al.*, 2008; COWI, 2014; Haines-Young and Potschin, 2014; Potschin and Haines-Young, 2011). This is done when attempting to complete a full ES assessment, including valuations, as to avoid double counting (Boyd and Banzhaf, 2007; COWI, 2014; Fisher *et al.*, 2009; MA, 2005; ONEMA, 2011; Potschin and Haines-Young, 2011; Potschin and Haines-Young, 2014; TEEB, 2010). Accordingly, context-specific definitions of final and intermediate ESs will have to be established when attempting to develop a conceptual model based on the flows and linkages between different ESs.

A further distinction needs to be made between biotic and abiotic dependent ESs (COWI, 2014; EC, 2013; Fisher *et al.*, 2009; TEEB, 2010). Some ES definitions consider ESs to be fundamentally dependent on living organisms and/or processes, therefore, all abiotic outputs from ecosystems are considered separately as part of ‘natural capital’ rather than ESs themselves (Costanza, 2008; COWI, 2014). On the other hand, the French National Agency for Water and Aquatic Environments (ONEMA, 2011) has differentiated, as previously mentioned, between ‘Environmental Services’ and ‘Ecological Services’, of which the former considers services both biotic and abiotic compartments (COWI, 2014).

Figure 3 is an adaptation of the ES cascade model, it illustrates the pathway from ecosystem structure and processes to human well-being as explained by The Economics of Ecosystems and Biodiversity (TEEB) report (De Groot *et al.*, 2010a; TEEB, 2010). This illustration graphically separates the elements of the cascade, which are components of ecosystems and biodiversity, human well-being and final ESs. Importantly, this model elucidates the impacts of feedbacks and management and/or restoration interventions. Figure 3 introduces negative feedbacks into the conceptual model and includes more relevant features of real world systems such as environmental pressures and mitigating policies (Braat and De Groot, 2012; De Groot *et al.*, 2010a). Positive feedbacks were included through institutions, judgements, restoration and management; thus, effectively connecting social and environmental aspects

of ESs (Braat and De Groot, 2012). This adaptation addresses the issue of how the ES cascade model implies a unidirectional downward flow from ecosystems to human well-being without any other forms of feedback or influences (Braat and De Groot, 2012; Salles, 2011).

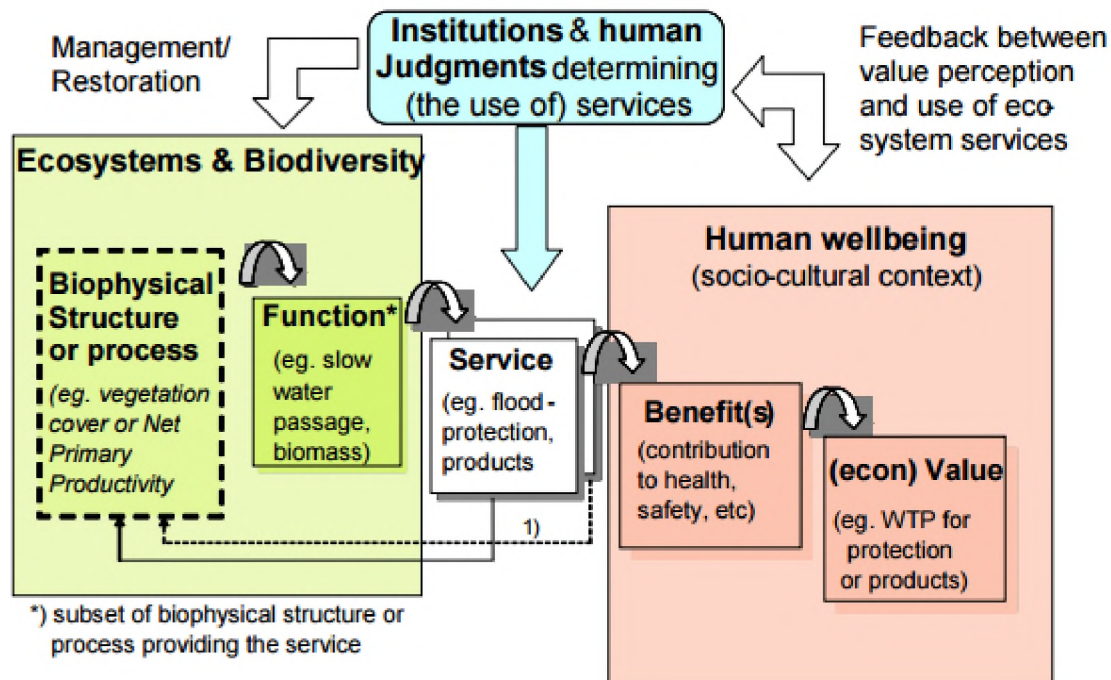


Figure 3: The pathway from ecosystem structure and processes to human well-being (De Groot *et al.*, 2010a).

The challenge of identifying and classifying ESs is conceptually and practically problematic as it requires a precise and commonly accepted definition of the term (Costanza, 2008; Haines-Young and Potschin, 2014; TEEB, 2010). This is because ESs are essentially ‘boundary objects’; they assist in transmitting and coordinating thinking between disciplines, despite the absence of a benchmark definition (Costanza, 2008; Haines-Young and Potschin, 2014). The concept is widely used due to it being vague and open to interpretation, thus any consensus regarding classification is difficult to realise (Haines-Young and Potschin, 2014). Ultimately, there are two primary definitional issues surrounding ESs that need to be addressed when identifying, classifying and assessing ESs within a particular decision context:

- Whether ESs are considered as benefits or as contributions towards human well-being (through benefits supported by ‘final’ ESs) (Costanza, 2008; Potschin and Haines-Young, 2011; Haines-Young and Potschin, 2014).

- Whether ESs are only considered as outputs dependent on living processes or if they include pure abiotic components (e.g. hydropower) (Haines-Young and Potschin, 2014).

2.3. Ecosystem Service Classification and Typology

The classification and/or typology of ESs is challenging technically and conceptually (Haines-Young and Potschin, 2014; Sokal, 1974). Furthermore, a commonly accepted and robust classification scheme is urgently required to facilitate various applications in policy and management decision-making and research (Costanza, 2008; Fisher *et al.*, 2009; Haines-Young and Potschin, 2014). Classification of ESs can be broadly categorised into three main international classification systems, namely: The Millennium Ecosystem Assessment (MA, 2005), The Economics of Ecosystems and Biodiversity study (TEEB, 2010) and The Common International Classification of Ecosystem Services (CICES) (COWI, 2014; Haines-Young and Potschin, 2013; MAES, 2012). These systems differ according to the aforementioned definitions of ESs and indicate how the 'Ecosystem Service Approach' has developed into a prominent tool for the operationalisation of environmental accounting protocols and detailing the values and benefits of ESs to society and decision-makers (COWI, 2014).

The MA (2005) popularised the concept of ESs and the basic classification of ESs into four broad categories: provisioning, regulating, supporting and cultural services (Boyd and Banzhaf, 2007; Costanza, 2008; EC, 2013; Fisher *et al.*, 2009; Haines-Young and Potschin, 2014; ONEMA, 2011; TEEB, 2010; Van Den Berg *et al.*, 2013; Van Wilgen *et al.*, 2008). Figure 4 illustrates the different categories of ESs, various drivers of change and how these factors affect human well-being (UN, 2008). This conceptual model illustrates the impact of direct and indirect drivers of change and ecosystem degradation as these have the potential to disrupt the natural flow of ESs highlighted in Figures 2 and 3.

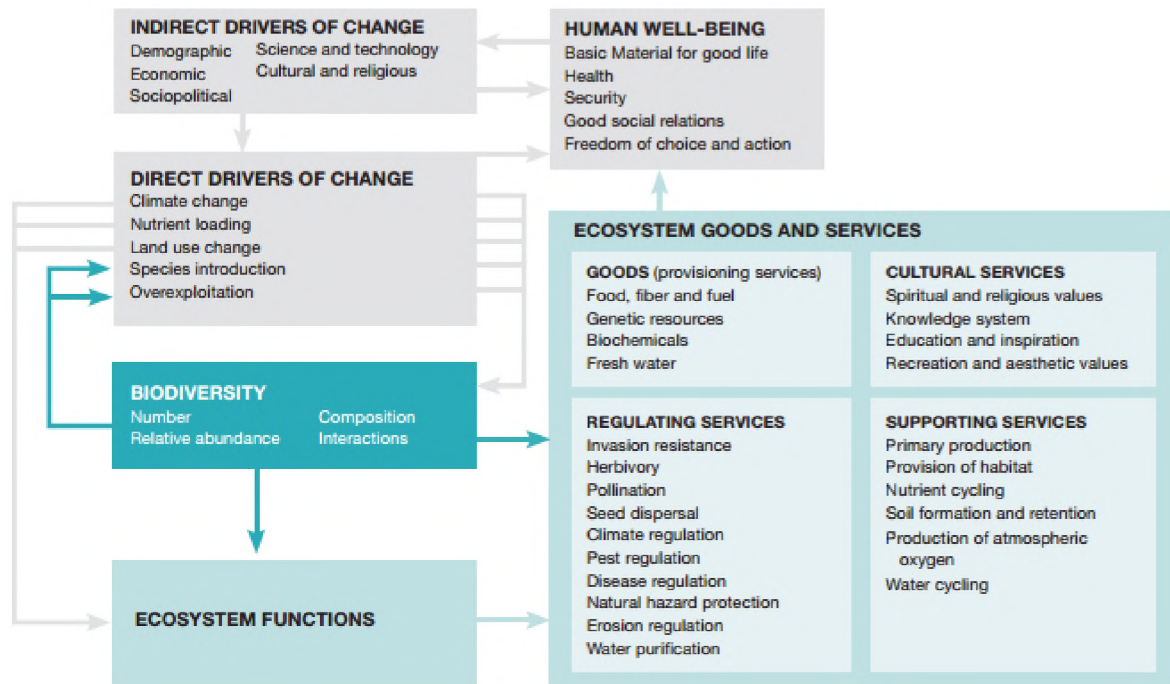


Figure 4: Biodiversity, ecosystem functioning and drivers of change (UN, 2008).

Provisioning services can be described as the material or energy outputs from an ecosystem such as water (COWI, 2014; MA, 2005; TEEB, 2010). Consumers and buyers of these services can influence their production via different market mechanisms, and governments via policy implementation and regulation (Van Den Berg *et al.*, 2013). Regulating or maintenance services are termed as the various ways biotic organisms moderate and/or mediate the surrounding environment, this includes services which directly or indirectly impact human well-being (UN, 2008; COWI, 2014; EC, 2013; MA, 2005; TEEB, 2010). In contrast, cultural services are considered as the non-material (usually non-consumptive) outputs from ecosystems that affect the mental and/or physical state of human well-being (COWI, 2014; EC, 2013; Ginsburg *et al.*, 2010; MA, 2005; TEEB, 2010; Van Den Berg *et al.*, 2013). These three categories of ES are considered to fall under the overarching paradigm of final ESs. Final ESs are identified as direct contributions that ecosystems make towards human well-being (Balmford *et al.*, 2008; COWI, 2014; De Groot *et al.*, 2010b; Haines-Young and Potschin, 2014; Potschin and Haines-Young, 2011). Thus, supporting services fall under the paradigm of intermediate ESs which encompass all the ecological/biological processes, structures and functions that underlie the provision of final ESs (Figure 2 and 3) (Costanza, 2008; COWI, 2014; Haines-Young and Potschin, 2011). The importance of ESs and their classification lies in

understanding and communicating the potential for human activities to degrade the capacity of ecosystems to maintain the supply of these essential services (Van Den Berg *et al.*, 2013).

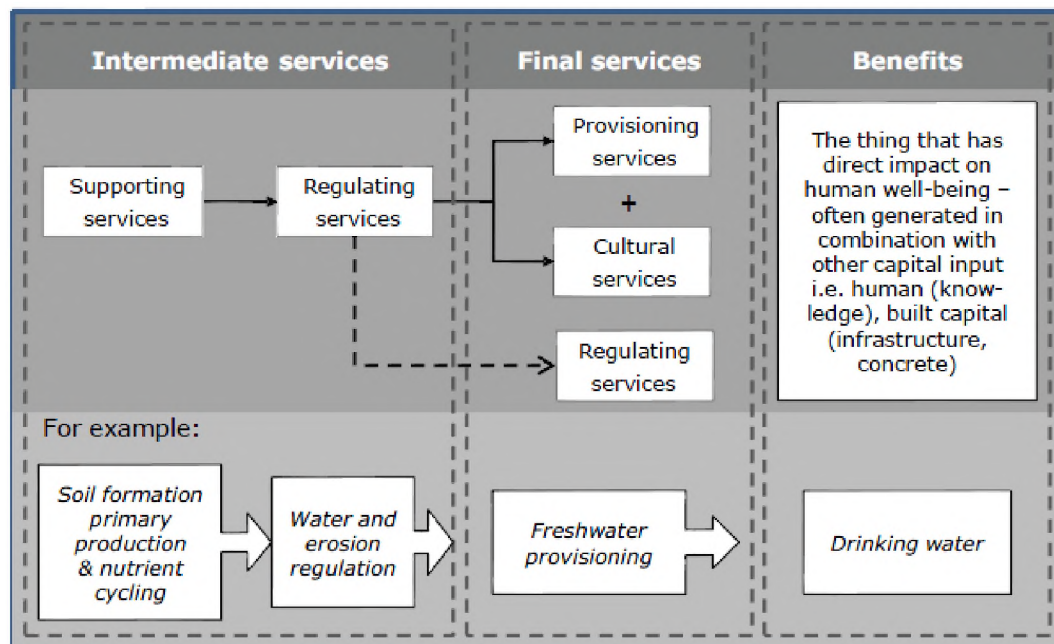


Figure 5: The distinction between intermediate services, final services and benefits (adapted from Fisher *et al.*, 2008) illustrated by the stylised relationship between supporting, regulating, provisioning and cultural services (Ginsburg *et al.*, 2010).

Figure 5 elucidates the relationship between the different MA (2005) categories of ESs, intermediate and final ESs and direct benefits associated with ESs by using the example of freshwater provision as a final ES.

The CICES and TEEB classification schemes attempt to extend the MA (2005) approach for numerous different applications; Table 1 compares the three classification schemes in terms of their individual ES categories. MAES (2012) describes how the MA classification is globally recognised and used for general sub-global assessments, while the TEEB classification is based on the original MA classification and is used within current TEEB studies throughout Europe. The CICES scheme offers a step-by-step hierarchal system of classification tailored towards environmental accounting at multiple scales of analysis and has been developed on the foundations of the other two classification schemes (EEA, 2015; EC, 2013; MAES, 2012).

Fisher *et al.* (2009) discuss several broad characteristics of ESs that can assist in the classification of ESs for a variety of decision-making contexts: the public-private good aspect,

spatial and temporal variation, joint production, complexity and benefit dependence. A thorough understanding of the key characteristics of ESs can contribute towards the development and improvement of effective management and policy applications (Brauman *et al.*, 2007; Fisher *et al.*, 2009).

Table 1: Ecosystem Service Categories in MA, TEEB and CICES (MAES, 2012).

MA categories	TEEB categories		CICES v4.3 group [†]
Food (fodder)	Food	Provisioning services	Biomass (Nutrition)
			Biomass (Materials from plants, algae and animals for agricultural use)
Fresh water	Water		Water (for drinking purposes) (Nutrition)
			Water (for non-drinking purposes) (Materials)
Fibre, timber	Raw Materials		Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
Genetic resources	Genetic resources		Biomass (genetic materials from all biota)
Biochemicals	Medicinal resources		Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
Ornamental resources	Ornamental resources		Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
			Biomass based energy sources
			Mechanical energy (animal based)
Air quality regulation	Air quality regulation	Regulating services (TEEB)	(Mediation of) gaseous/air flows
Water purification and water treatment	Waste treatment (water purification)		Mediation (of waste, toxics and other nuisances) by biota
			Mediation (of waste, toxics and other nuisances) by ecosystems
Water regulation	Regulation of water flows Moderation of extreme events		(Mediation of) liquid flows
Erosion regulation	Erosion prevention		(Mediation of) mass flows
Climate regulation	Climate regulation		Atmospheric composition and climate regulation
Soil formation (supporting service)	Maintenance of soil fertility		Soil formation and composition
Pollination	Pollination		Lifecycle maintenance, habitat and gene pool protection
Pest regulation	Biological control		Pest and disease control
Disease regulation			Lifecycle maintenance, habitat and gene pool protection
Primary production	Maintenance of life cycles of migratory species (incl. nursery service)	Regulating and supporting services (MA)	Soil formation and composition
Nutrient cycling (supporting services)	Maintenance of genetic diversity (especially in gene pool protection)		(Maintenance of) water conditions
			Lifecycle maintenance, habitat and gene pool protection
Spiritual and religious values	Spiritual experience	Regulating and maintenance services (CICES)	Spiritual and/or emblematic
Aesthetic values	Aesthetic information		Intellectual and representational interactions
			Intellectual and representational interactions
Cultural diversity	Inspiration for culture, art and design		Spiritual and/or emblematic
Recreation and ecotourism	Recreation and tourism		Physical and experiential interactions
Knowledge systems and educational values	Information for cognitive development		Intellectual and representational interactions
			Other cultural outputs (existence, bequest)

The public-private good aspect involves how different ESs will influence and react to market forces based on whether they are rival/non-rival and/or excludable/non-excludable (Jack *et al.*, 2008; ONEMA, 2011). If a good or service is considered rival then its utilisation will

diminish the availability of that service to other users and *vice versa* (ONEMA, 2011; Van Den Berg *et al.*, 2013). If a good/service is excludable then the 'owner'/'user' can keep others from using/consuming that particular good/service and *vice versa* (Fisher *et al.*, 2009; Jack *et al.*, 2008). Hence, private goods are considered to be rival and excludable while public goods are considered to be non-rival and non-excludable. Figure 6 illustrates this differentiation and where ESs fit into the spectrum from excludable to non-excludable and rival to non-rival. ESs can fall anywhere within this spectrum, however, most ESs are considered to be public goods, but there are also many circumstances where they behave as private, open access and/or club goods (Figure 6) (Fisher *et al.*, 2009; ONEMA, 2011; Van Den Berg *et al.*, 2013). Furthermore, understanding where and how ESs fit into the public-private goods spectrum is complex as it is a function of individual ES dynamics as well as the various social systems that interact with these services (De Groot *et al.*, 2010a; Fisher *et al.*, 2009).

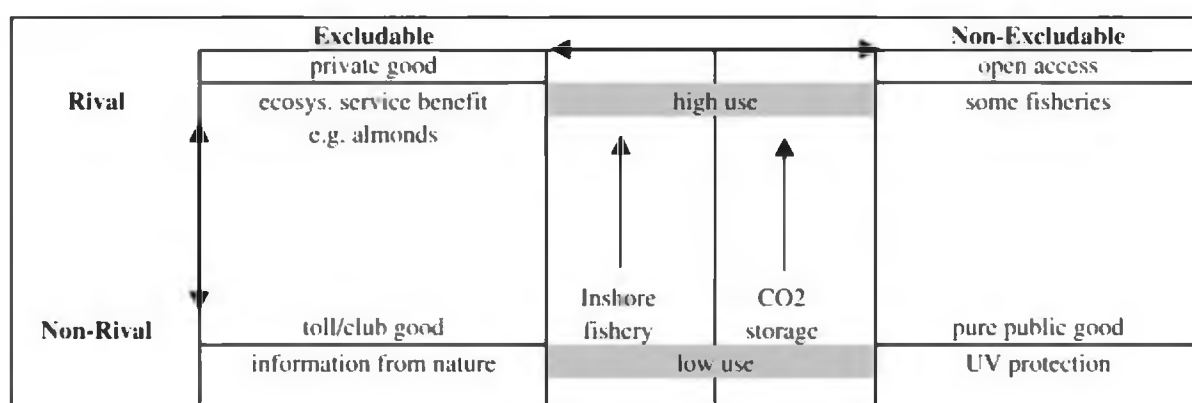


Figure 6: Good and service characterisation continuum from rival to non-rival and excludable to non-excludable (Fisher *et al.*, 2009).

ESs are heterogeneous in space in time, understanding this notion can assist in their classification (Fisher *et al.*, 2009; Hein *et al.*, 2006; Limburg *et al.*, 2002). For example, mountain top ecosystems can provide water regulation services and benefits in the form of extended water provision to downslope communities (Fisher *et al.*, 2009). The concept of joint production is that ESs can deliver multiple benefits for human well-being and/or discrete ecosystems can provide several ESs (Daily, 1997; Fisher *et al.*, 2009). The notions of positive and negative economic externalities play a pivotal role in understanding linkages between ESs and human welfare. Crocker and Tschirhart (1992), Gómez-Baggethun *et al.* (2010) and Fisher *et al.* (2009) outline the concept of economic externalities in terms of ESs as the inadvertent

increase (positive) or decrease (negative) in the provision of a service brought about by the action of another agent in the system. For example, one person's abstraction of water in a catchment may preclude another's benefit from aquatic ecosystem health downstream due to declining water volume and habitat. The diverse array of impacts that human agency has on the environment and *vice versa* is challenging to evaluate since positive and negative externalities occur throughout complex socio-ecological systems (Crocker and Tschirhart 1992). Dynamic modelling is being increasingly utilised to assess the unintended consequences that result from these interlinked systems (Boumans *et al.*, 2002; Fisher *et al.*, 2009). These characteristics of ESs are important considerations when determining classification and identification schemes for model development within particular decision-making contexts (Fisher *et al.*, 2009).

Analysing complex relationships between process, structure and service is made more challenging due to ecosystems displaying properties of non-linearity, time lags and feedbacks (Limburg *et al.*, 2002). However, this characteristic could potentially assist in the classification of ESs as it allows for some degree of simplification and generalisation, which can be tailored towards particular outcomes and applications (De Groot *et al.*, 2010a; EC, 2013; Fisher *et al.*, 2009). Boyd and Banzhaf (2007) outline the concept of benefit dependence; desired benefits will dictate how an ES is understood. Whether an ES is considered as 'final' or 'intermediate' will depend on the beneficiaries as well as what benefit is valued, monitored, modelled or measured (Boyd, 2007; Fisher *et al.*, 2009).

Stakeholders perceive different benefits from identical ecosystem processes and thus, they can be conflicting (Hein *et al.*, 2006; Turner *et al.*, 2003). Fisher *et al.* (2009) describe the example of water regulation being considered as an intermediate input into the final service of clean water provision. However, if the desired service was fish production then clean water provision would be considered an intermediate input as opposed to final service. Ultimately, Fisher *et al.* (2009: 644) argue that classification systems should be underpinned by: "1) a clear and robust definition 2) the characteristics of the ecosystem or ecosystem services under investigation and 3) the decision context or motivation for which ecosystem services are being considered".

2.4. Ecosystem Service Identification

ESs need to be accurately described and measured before they can be valued, assessed, mapped or included in any form of modelling or accounting (Haines-Young and Potschin, 2013). The identification of relevant, context-specific ESs is the first step in any assessment process (COWI, 2014; EC, 2013; MA, 2005; TEEB, 2010). The identification phase of ES assessment begins with the analysis and comparison of a list which classifies all relevant ESs (e.g. CICES, TEEB, MA), with sufficient information about the ecosystem/catchment at hand (COWI, 2014). The second phase employs a transdisciplinary approach towards participatory processes (such as integrated professional workshops or interviews) that facilitate the involvement of all relevant beneficiary/stakeholder groups (Costanza, 2008; COWI, 2014; Fisher *et al.*, 2009; Saladin *et al.*, 2012).

Saladin *et al.* (2012) describe an approach towards identifying ESs, which include an initial literature review and consultation of water management and ecosystem experts; this produced a qualitative description of the ESs under study as well as a degree of categorisation. Such stakeholder interaction can often produce highly valuable information that is not available in the literature and goes beyond the issues focused on by specialists (COWI, 2014). When there is a need for supplementary data after the literature review, expert consultation and/or participatory process, the last step in the identification of ESs may involve an on-site examination, additional mapping and/or local expert judgement (COWI, 2014). As ESs are highly case- and site-specific, there is always the potential for a lack of data and information to exist, thus, not allowing all ESs to be accounted for (COWI, 2014; Saladin *et al.*, 2012). Therefore, a step-wise process, even without a complete list of ESs, may still provide some form of value in the communication and decision-making processes (Cardoso *et al.*, 2013; ONEMA, 2011). There are numerous contrasting methods towards ES assessment, valuation, mapping and modelling, however, ES identification and classification are consistently identified within the first major phase in any of these approaches (Bagstad *et al.*, 2013).

2.5. Ecosystem Service Assessment, Management and Sustainability Tools

The conception of ESs and the abundance of scientific and socio-economic research that has contributed to the development of this transdisciplinary field of inquiry has generally been founded on the core principle of sustainability. Decision-makers are constantly faced with the

trade-off between simultaneously promoting economic development that meets the needs of the people and ensuring the functionality and productivity of crucial ESs (Nahman *et al.*, 2009). Recent research has highlighted the widespread decline in ES provision as a result of their unsustainable use and human driven ecosystem degradation (Bagstad *et al.*, 2013; Boumans *et al.*, 2015; De Groot *et al.*, 2010b; Egoh *et al.*, 2008; Goodstein, 2011; MA, 2005; Nunes *et al.*, 2014; WRI, 2000). Thus, to secure the current and future provision of these crucial services, areas that maintain the various ecosystem components, processes and functions need to be effectively and sustainably managed (Chan *et al.*, 2006; Egoh *et al.*, 2008; Fernald *et al.*, 2012; Hanley *et al.*, 2007; Van Jaarsveld *et al.*, 2005). Several of the most widely-used and relevant (in relation to this study) approaches for ES assessment and decision-support frameworks will be presented and discussed, as well as a general discussion on the notion of sustainability of ESs, payment for ESs and complexity theory.

Villa *et al.* (2014) and Bagstad *et al.* (2013) highlight the most prevalent and significant shortcomings of current ES-driven mapping, measurement, valuation and assessment models. Current research methodologies still fall short in their ability to fully incorporate the complex, multiscale, socio-economic and biophysical dynamics associated with ESs; while practical application remains tedious and data-intensive due to specific model parameterisation requirements (McComb *et al.*, 2006; Villa *et al.*, 2014). In recent time it has become increasingly clear that any overriding, “one model fits all” paradigm is generally not appropriate to assess the diversity of complex socio-economic scenarios (Villa *et al.*, 2014). Effective sustainable management of complex socio-ecological systems requires a suite of decision-support tools and assessment models that are geared towards informing relevant policy and management decisions (Bagstad *et al.*, 2013; Grigg *et al.*, 2009; Midgley *et al.*, 2012).

The Artificial Intelligence for Ecosystem Services (ARIES) tool is an integrated ES modelling instrument and methodology that incorporates a uniform conceptualisation of ESs that attempts to place equal emphasis on the flow, production and use of ESs by society (Vigerstol and Aukema, 2011; Villa *et al.*, 2014). The framework is open source and facilitates quantitative mapping of ES flows at the watershed or landscape scale (Bagstad *et al.*, 2013; Villa *et al.*, 2011). ARIES uses a large variety of spatial data as input into models that encode ecological production functions into probabilistic and deterministic models (Villa *et al.*, 2014).

Additionally, it is one of the most well-known, dynamic and current of the generalizable, public-domain ES assessment tools available (Bagstad *et al.*, 2013; Vigerstol and Aukema, 2011; Villa *et al.*, 2014). Importantly, ARIES attempts to address the challenge of modelling multiple application contexts by incorporating integration technologies that allow for the development and inclusion of numerous customisable models (Villa *et al.*, 2014). Villa *et al.* (2014) provides a detailed account of the development, scale of application, advantages and disadvantages of the ARIES tool.

Along with ARIES, the Multiscale Integrated Model of Ecosystem Services (MIMES) are probably the most-widely used and known public-domain ES assessment tools (Bagstad *et al.*, 2013). MIMES is an open source, analytical, GIS-based (geographic information systems) framework developed to analyse different dynamics associated with ES function and human activity (Boumans *et al.*, 2015). MIMES requires access to Simile modelling software, declarative software designed to develop spatially explicit system dynamics models (Muetzelfeldt and Massheder, 2003). MIMES combine different modes of quantitative information to illustrate how benefits derived from ESs are increased or decreased; the synthesised information produces alternate scenario forecasts of ESs and human-use dynamics (Boumans *et al.*, 2015). MIMES necessitates the specification of multiple human and ecological dynamics (Boumans *et al.*, 2015). Depending on the context, this can be highly data-intensive and onerous, but it facilitates the assessment of the effects of different ES scenarios at varying spatial and temporal scales (Bagstad *et al.*, 2013). Boumans *et al.* (2015) provides a detailed explanation of the method development, technical software, model applications, advantages and disadvantages of MIMES.

Bayesian belief networks (BBNs) are multivariate probabilistic statistical models that graphically depict a set of variables and their conditional dependencies or drivers (Jenson, 2001; Landuyt *et al.*, 2013). BBNs use directed acyclic graphs to illustrate various model interdependencies, as well as conditional probability tables to represent the relative strengths of the linkages between variables (Aguilera *et al.*, 2010). Importantly for ES assessment and modelling, BBNs are able to account for system uncertainty through incorporating both qualitative expert knowledge and quantitative data (Landuyt *et al.*, 2013). However, BBNs are severely limited in their ability to dynamically model complex systems (e.g. aquatic ecosystems) because of the absence of feedback loops and exclusive use of

discrete variables (Landuyt *et al.*, 2013; Zhang and Govindaraju, 2000). A detailed description of BBNs is provided by Jensen (2001), Landuyt *et al.* (2013) provides an intricate breakdown of the use of BBNs in ES assessment and modelling.

The Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) tool is a collection of open source models used to map and value a chosen suite of ESs with the aim of identifying and highlighting patterns change in ES provision that result from anthropogenic activity (Daily *et al.*, 2009; Tallis and Polasky, 2009; Vigerstol and Aukema, 2011). The current toolset combines 18 individual GIS and production function based ES models that are tailored to various landscapes (Vigerstol and Aukema, 2011). The various models allow for multi-scale application, however, large amounts of data are necessary to produce sufficiently accurate results with some of the tools simplifying ecological and hydrological processes (Daily *et al.*, 2009; Vigerstol and Aukema, 2011). The InVEST tool is a product of the Natural Capital Project, a detailed description of the various applications, strengths, weaknesses and current research endeavours presented by NCP (2016).

The Soil and Water Assessment Tool (SWAT) and Variable Infiltration Capacity (VIC) methods were developed for general hydrological ecosystem assessment, and have been used as a hydrological ES assessment tool in a variety of contexts (Vigerstol and Aukema, 2011). SWAT attempts to model water quantity and quality on a daily scale with the overall aim of evaluating the effect of land-use changes on catchment yield, agricultural pollutants and sediment (Arnold *et al.*, 1998; Arnold and Fohrer, 2005). VIC is a gridded hydrological model that analyses and estimates water movement through precipitation, runoff and infiltration at a large scale; stream flow and water yield are the primary variables (Liang *et al.*, 1994; Nijssen *et al.*, 1997). Both of these tools are designed purely for hydrological analyses, thus, both require hydrological expertise to use and detailed data requirements to generate outputs with acceptable resolutions (Vigerstol and Aukema, 2011).

Vigerstol and Aukema (2011) discuss and compare the ARIES, InVEST, SWAT and VIC tools in the context of modelling freshwater ESs. Bagstad *et al.* (2013) provide a detailed analysis of 17 different ES decision-support tools and rates their individual performance against eight evaluative criteria that determine each tool's appropriateness and readiness for widespread application in private and public sector decision-making.

Software based decision-support modelling tools are not the only methods used to manage ESs; decision and management frameworks that utilise multiple models and/or a combination of qualitative and quantitative inputs can provide simpler and more holistic options. Nonetheless, these two general approaches often complement each other when used in unison. Grigg *et al.* (2009) present the Ecosystem Services Benchmark (ESB) tool, which was designed to support institutional investor decision-making around the opportunities and risks faced by businesses resulting from their dependencies and impacts on ESs. The ESB is specifically focused on analysing agricultural supply chains (but also has broader natural resources applicability) and the impacts and dependence on biodiversity and ESs related to the harvesting and production of raw materials (Grigg *et al.*, 2009). The approach allows for objective examination of comparative risk exposure associated with ESs, which provides a solid platform for stakeholder engagement; ultimately, aiming to improve company profitability and investment decision-making into ES delivery (Grigg *et al.*, 2009).

The System of Environmental-Economic Accounting (SEEA), developed by the European Commission (EC, 2013), is an experimental ecosystem accounting approach that integrates statistical frameworks for organising and analysing biophysical data to measure and record changes in ESs and assets. Importantly, the SEEA attempts to link all information to economic and other human activity to create accountability. However, the approach has been critiqued surrounding issues of double counting and the valuation of intricate ESs (Grizzetti *et al.*, 2015; Pascual *et al.*, 2010). The SEEA defines ecosystem stocks (assets) and flows (ESs) in relation to a range of other economic, environmental and social factors to facilitate an account of the inputs and outputs associated with ESs (EC, 2013)

A policy decision-support framework focusing on the dual role of ES assessment in water management is put forward by ONEMA (2011). The three assessment stages are “analysis and identification of the situation; identification of the best measures for resolution or preservation; economic assessment and political choice” (ONEMA, 2011: 35). This framework facilitates an initial appraisal of the scenario or human-environment interactions that allows for the identification of potentially conflicting situations, thus, the determination of scale and setting to guide action towards a desired goal (ONEMA, 2011). Then an assessment of potential technical interventions and possible outcomes is conducted that lays the foundation

for an economic assessment, effectively completing the diagnosis by excluding any interventions that would sustain disproportionate costs to benefits (ONEMA, 2011).

Pollard *et al.* (2014) present an integrative framework designed for context based decision-making in natural resource management. The framework comprises of ten key steps (Pollard *et al.*, 2014: 67) and focuses on reframing catchments as recognisable complex socio-ecological systems, allowing for the understanding and incorporation of system resilience in the face of rapid system changes (Pollard *et al.*, 2014). This is largely due to the integration of the VSTEPP paradigm, a heuristic encouraging a complete inclusion of influences: values, social, technological, ecological, economic, and political (Biggs and Rogers, 2003). In a similar vein, De Groot *et al.* (2002) describe a general framework for integrated ES assessment and valuation which attempts to unpack the concept of ESs to provide a more holistic assessment of complex systems. Coarse distinctions are made between the concepts of ecosystem processes, functions and services to avoid conceptual and practical overlapping (De Groot *et al.*, 2002). Recognised ES valuation methods are used to describe the linkages between these different concepts and contribute towards a better understanding of ES value and ultimately better decision-making (De Groot *et al.*, 2002).

De Groot *et al.* (2010b) and Gallopin (2003) emphasise that most challenges facing the integration of ES management into conventional landscape planning, management and policy decision-making are due to the intrinsic complexity of socio-economic systems and contrasting ideas surrounding the meaning and implications of sustainability (such as the use of different valuation criteria functions). The concepts of sustainability, sustainable development and sustainable utilisation can be considered from an anthropocentric or ecocentric viewpoint because ESs provide benefits directly and indirectly to people (DWA, 2013b; Fernald *et al.*, 2012; Gallopin, 2003). This is closely linked to the notions of 'weak' and 'strong' sustainability. Solow (1993) and Hartwick (1978) introduce these concepts and highlight how weak sustainability incorporates the idea that human capital (e.g. labour, infrastructure etc.) can substitute for natural capital (e.g. stock of environmental assets); while strong sustainability assumes that these two types of capital are complimentary, as opposed to being interchangeable. When investigating ESs, the notions of sustainability and sustainable management/utilisation are considered from an anthropocentric viewpoint as ESs provide benefits directly to people.

Modern ES management is underpinned by the notion of sustainability, however, linear and mechanistic science has not been able to cope with issues surrounding sustainability due to the complex and self-organising nature of these vulnerable systems (Hjorth and Bagheri, 2006; Limburg *et al.*, 2002; Manuel-Navarrete *et al.*, 2007; Nahman *et al.*, 2009). Gallopin (2003) explains how sustainability is not a fixed state of constancy but rather a dynamic system property that preserves the essence of the system amidst constant change. Complexity theory was developed through a critique of reductionist and linear scientific practice in coupled human-environment systems (Pollard *et al.*, 2008). Recent research has highlighted the failure of conventional, linear thinking in progressing the sustainability agenda, in many cases even exacerbating the problem (Walker and Salt, 2006). Emerging schools of thought encourage systemic thinking and cross-disciplinarity (non-linear and organic) as a more comprehensive approach towards understanding the sources of and solutions to modern, complex problems such as ES decline (Hjorth and Bagheri, 2006; Palmer *et al.*, 2014; Palmer *et al.*, 2004; Pollard *et al.*, 2008; Walker and Salt, 2006).

Payments for ESs or payments for environmental services (PES) is a widely acknowledged and researched approach aimed at promoting the sustainability of ESs through financial incentivisation. PES is defined by Wunder (2007: 50) as “a voluntary, conditional agreement between at least one ‘seller’ and one ‘buyer’ over a well-defined environmental service - or a land use presumed to produce that service”. PES agreements and policies aim to compensate individuals or communities for engaging in specific activities that increase the provision of certain ESs, these include everything from water purification and flood mitigation to carbon sequestration (Jack *et al.*, 2008; Mander *et al.*, 2010). Wunder (2005) outlines five criteria that are applied in a working prescriptive definition of the concept of PES: (1) a voluntary transaction where (2) a well-defined environmental good or service (3) is “bought” by a buyer (4) from a provider (5) so that the provider continues to secure the provision of the service into the future.

However, in recent times PES schemes have been found to be practically problematic, being largely dependent on having ideal pre-conditions and being fraught with issues around ethics, equity, efficiency and efficacy (Alpizar *et al.*, 2007; Pirard *et al.*, 2010; Van Den Berg *et al.*, 2013; Wunder, 2007). PES initiatives depend on incentivising individuals to change their behaviour in one way or another and thus, form part of the market- or incentive-based

mechanisms of environmental policy (Jack *et al.*, 2008). These types of approaches are akin to directly paying individuals and/or communities in return for environmental conservation efforts, more commonly known as conservation contracting (Ferraro, 2000). A large proportion of people in Southern Africa rely heavily on natural resources and ecosystems for their livelihoods. The overarching aim of PES and conservation contracting schemes is to promote resilient livelihoods in the face of climate and disaster risk in an attempt to create win-win situations where key ESs are restored to improve sustainability in the long run (Mander *et al.*, 2010; Midgley *et al.*, 2012; Van Den Berg *et al.*, 2013).

For the purpose of this study, sustainable utilisation will be defined as the use of ESs at levels and in ways that allow them to continue renewing themselves for all practical purposes indefinitely (Adapted from IUCN (1980)). This conception attempts to facilitate a compromise between the two competing views of sustainability by stating ‘for all practical purposes indefinitely’. Such ‘practical purposes’ could be considered for human and/or environmental benefit. However, due to the inherent complexity and interconnectedness of these systems, the benefits of improving an ecosystem’s ability to provide services will more often than not directly or indirectly benefit both parties (De Groot *et al.*, 2010b; Limburg *et al.*, 2002; Vihervaara *et al.*, 2010). It is clear that attempting to incorporate sustainable ES management into policy and decision-making requires a systemic approach with clear system specifications to which the concept of sustainability is applied (Gallopín, 2003; Limburg *et al.*, 2002; Moffatt and Hanley, 2001).

2.6. Aquatic Ecosystem Services

Freshwater ESs derived from aquatic ecosystems are essential to human survival and are declining on a global scale (Brauman *et al.*, 2007; Limburg, 2009; Van Jaarsveld *et al.*, 2005). Aquatic ecosystems are highly sensitive and dynamic environments, where small changes can lead to severe system-scale effects, making management of these environments challenging (Brauman *et al.*, 2007; Bunn and Arthington, 2002; Guswa *et al.*, 2014; Kotze *et al.*, 2008; Limburg, 2009).

Considering the previous discussion focusing on defining, identifying and classifying ESs for their integration into modelling and decision-support tools/frameworks, a clear and robust definition of AESs is critical. Such definitions will allow for the selection of an appropriate

classification scheme for the purpose of this research and ultimately help address aims and objectives of the study in the specific decision context.

To address the primary research question and associated sub-goals of this project, final and intermediate AESs are defined below.

Final AESs:

Biophysical functions, processes or conditions of aquatic ecosystems that are utilised (actively or passively) to provide benefits that directly contribute towards the welfare of one or more human beneficiary.

Intermediate AESs:

Biophysical functions, processes or conditions of aquatic ecosystems that only provide benefits to humans indirectly through effects on other final ecosystem services first.

These definitions extend those proposed by Boyd and Banzhaf (2007), Fisher *et al.* (2009), Johnston and Russell (2011) and Limburg (2009). Final AESs only include provisioning, cultural and certain regulating services that directly contribute towards human welfare. While intermediate AESs could fall under any of the four ES categories outlined by the MA (2005). The concepts of ecosystem goods and services are both considered to fall under the term 'service'.

Aquatic ecosystem processes, functions and conditions include all abiotic and biotic components of ESs, as the definition of ecosystem described by the UN (1992) underpins the aquatic ecosystem component of this definition. As mentioned by EC (2013) and ONEMA (2011), these definitions do not need to distinguish between 'ecological services' and AESs as the focus of the study is only on one ecosystem type. Under this definition, aquatic ecosystems include all freshwater rivers, streams, lakes, ponds, groundwater and any directly associated abiotic and biotic components within the delineated study area (Landers and Nahlik, 2013). AESs derived from wetlands are excluded from this definition due to the methodical and conceptual issues related to delimitation of wetland-based aquatic ecosystems explained by ONEMA (2011: 74). However, wetlands are included in the causal

analysis of AESs and construction of the CLD as they play an integral role in the hydrodynamics and complexity of aquatic ecosystems.

Ecohydrologic process (what the ecosystem does)		Hydrologic attribute (direct effect of the ecosystem)	Hydrologic service (what the beneficiary receives)
Local climate interactions Water use by plants	→	Quantity (surface and ground water storage and flow)	<u>Diverted water supply:</u> Water for municipal, agricultural, commercial, industrial, thermoelectric power generation uses <u>In situ water supply:</u> Water for hydropower, recreation, transportation, supply of fish and other freshwater products <u>Water damage mitigation:</u> Reduction of flood damage, dryland salinization, saltwater intrusion, sedimentation <u>Spiritual and aesthetic:</u> Provision of religious, educational, tourism values <u>Supporting:</u> Water and nutrients to support vital estuaries and other habitats, preservation of options
Environmental filtration Soil stabilization Chemical and biological additions/subtractions	→	Quality (pathogens, nutrients, salinity, sediment)	
Soil development Ground surface modification Surface flow path alteration River bank development	→	Location (ground/surface, up/downstream, in/out of channel)	
Control of flow speed Short and long-term water storage Seasonality of water use	→	Timing (peak flows, base flows, velocity)	

Figure 7: Relationship of hydrologic ecosystem processes to hydrologic services (Brauman *et al.*, 2007).

Brauman *et al.* (2007) describes how hydrologic services (analogous to final AESs) have differing attributes of quantity, quality, location and timing. These attributes are illustrated in Figure 7, which outlines how associated ecohydrologic processes (analogous to intermediate AESs) can impact any of the four attributes. Understanding how intermediate and final AESs affect one another is imperative when attempting to develop systemic, decision-support tools and models. These AESs are highly interdependent at different levels, understanding complex trade-offs between these services will help to identify whether the effects on one another are synergistic or competitive (analogous to positive and negative causal relationships described in section 3.2.1) (Brauman *et al.*, 2007; Limburg, 2009). Different scenarios of quantity, quality, location and timing of flow will improve some services at the expense of others

(depending on the nature of the value placed on each service by a particular beneficiary) (Brauman *et al.*, 2007).

This model of interlinked ESs that contribute to the generation of final ESs and ultimately provide some benefit to human well-being is illustrated through the concept of the ES cascade (Figures 2 and 3). Carvalho-Santos *et al.* (2014) present a further adaption of the ES cascade specifically for forest based AESs (Figure 8). The conceptual framework illustrates some of the most important biophysical functions that occur between water and forest systems as well as the resulting contributions towards human well-being. Water supply is demonstrated as one of the primary final AESs derived from the interface between these two natural systems, numerous intermediate AESs that contribute to the generation of this final service are illustrated (e.g. evaporation and geological substrate). Understanding the flows and linkages between various intermediate and final AESs and their provision benefits is integral to developing a model or decision-support framework based on AES value chains.

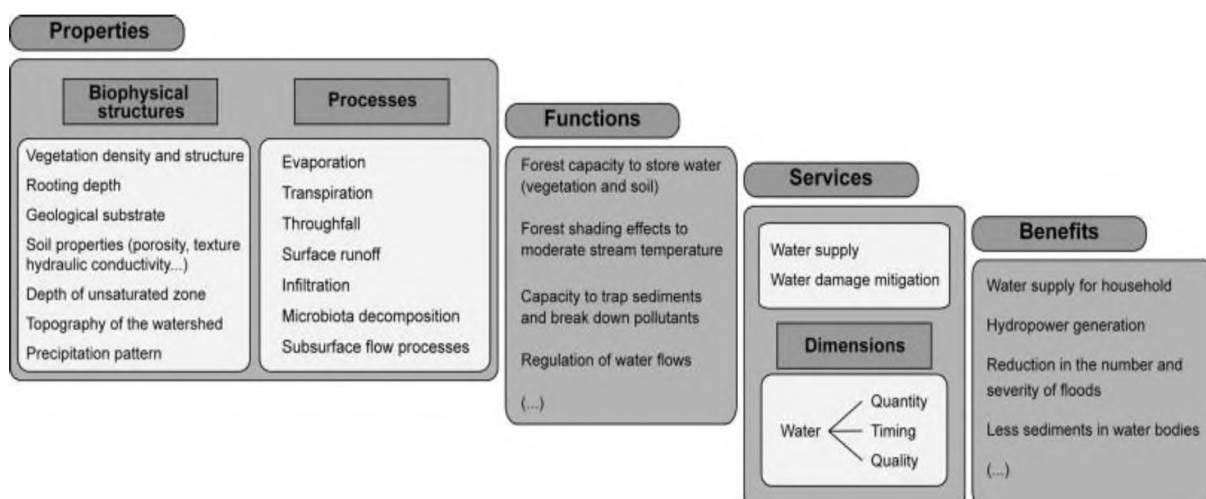


Figure 8: Conceptual ecosystem service cascade framework for hydrological services provision by forests (Carvalho-Santos *et al.*, 2014).

2.7. The Concept of Value and Valuation of Ecosystem Services

Since environmental goods and services (e.g. water) do not generally enter markets, or do so only imperfectly, market prices for these goods and services do either not exist, or inadequately capture their true value (Dixon and Pagiola, 1998). Thus, the monetary value of these benefits cannot be measured based on an existing market price. In such cases, it is necessary to conduct an economic valuation exercise using one (or more) of a number of non-

market valuation techniques. Efforts are being made to inform policy and decision makers and improve ES management by assigning them a value, effectively making them more relevant and visible (Van Den Berg *et al.*, 2013). Economic valuation involves placing a monetary value on the (often intangible) costs incurred or benefits derived by society as a result of the environmental or social impacts associated with, for example, water pollution (or, with a particular water pollution mitigation measure).

Values of ecosystems are broadly separated into three types: ecological, socio-cultural and economic value (De Groot *et al.*, 2002; Farber *et al.*, 2002; Limburg *et al.*, 2002). Although flows of ESs provide a nearly limitless set of valuable properties, a large proportion of their services remain unpriced through traditional markets (Hanley *et al.*, 2007). Inclusive valuations of ESs have not been notably successful due to a myriad of methodological challenges and because it is not always possible to accurately identify marketable value attributes (Barbier, 2011; Limburg, 2009; Parks and Gowdy, 2013). Thus, such goods and services are left without a market value, albeit not without value (Alpizar *et al.*, 2007; Ferraro, 2000). Up until now one way of accounting for such ESs was to present them as 'intermediate' services to 'final' services (i.e. goods and services which contain marketable value attributes) and then derive the value of the intermediate good or service from the marketable good or service by means of known valuation methods (Pascual *et al.*, 2010). This valuation technique helps to prevent double counting of intermediate and final ESs as well as ecosystem functions (Boyd and Banzhaf, 2007; Fisher *et al.*, 2009).

For the purpose of this study, the definition of the term "value" is not constrained to chrematistics (the study of market price formation for the purpose of making money (Martinez-Alier, 2005)) or exchange value in a market economy, because economists have used this narrow interpretation supported by precise (albeit limiting) mathematical frameworks to present the contributions of ecosystems to human well-being for far too long (Parks and Gowdy, 2013). Current controversies in applying valuation techniques derived from this narrow interpretation have exposed serious flaws in standard welfare economics (Kahneman, 2003a; Kahneman, 2003b). For example, the assumption that social values can be derived from the revealed or stated preferences of self-regarding narrowly-rational individuals (Jack *et al.*, 2008; Ryan and Spash, 2011). While the fields of behavioural psychology, neuroscience and social anthropology have shown that human decision-making

processes are also social and not only an individual process (Dawnay and Shah, 2005; Fischer and Hanley, 2007; Gilg and Barr, 2006; Hepburn et al., 2010; Sen, 1977). Hanley *et al.* (2007) describe how the typical economist's definition of value is based on ideals of rationality and consumer sovereignty, which provides the basis for economic valuation of ESs.

Turpie *et al.* (2010) describe economic value as the most a person is willing to relinquish in terms of other goods and/or services to obtain a desired good/service or state of the world. Money, being the universally accepted measure of economic value, allows one to determine the amount someone is willing to pay for a particular good or service, which then reveals how much of all other goods and services they are willing to sacrifice in turn (Farber *et al.*, 2002; Turner *et al.*, 1994; Turpie *et al.*, 2010). However, market prices do not always accurately illustrate economic value because many individuals are willing to pay more than the required market price for essential goods and services.

Economic systems and environmental ecosystems share many attributes; however, valuation of ESs has traditionally been driven by short-term human preferences (Limburg *et al.*, 2002). Limburg *et al.* (2002) argue that ES valuation needs to move from evaluating trade-offs amongst environmental resources to valuing the avoidance of dramatic ecosystem change. Stahel (2005) describes the inherent weaknesses of conventional value theory and highlights the relational and emergent characteristics of value focusing on how ecological and economic value of specific ESs have to be analysed within their individual spatiotemporal context. The application of a narrow interpretation of value does not present the full realm of values provided by ecosystems and neoclassical welfare economics is becoming increasingly recognised as limited in terms of reflecting this value (Goodstein, 2011; O'Neill et al., 2010). Costanza *et al.* (1998) and Farber *et al.* (2002) emphasise the differences between economic and ecological understandings of value and argue how, due to the existence of potential win-win situations for human activities within the natural environment, valuation of ESs is imperative to advance global ecological-economic system thinking and understanding. The concept of value must also be distinguished from the firm/individual perspective and the government perspective (Turpie *et al.*, 2010). Alternative approaches include behavioural economics (Mullainathan and Thaler, 2000), non-linear complexity theory, evolutionary economics, new-institutional economics, post-Keynesian economics, and neuro-economics.

2.7.1. Evaluating Aquatic Ecosystem Services in South Africa

Ginsburg *et al.* (2010) describe a simplified version of the South African Water Research Commission (WRC) framework for AES evaluation as illustrated in Figure 9. The four-phased approach includes (1) systems analysis, (2) assessing ecological change, (3) valuation of ESs and (4) evaluation of trade-offs. This framework attempts to link changes to ecosystems from the initial systems analysis resulting from differing management outcomes, to variations in the provision of ESs (Ginsburg *et al.*, 2010). The framework considers trade-offs between different water management scenarios by using economic valuations of the ESs (Ginsburg *et al.*, 2010). During the valuation of ESs phase, a rigorous process of selecting the appropriate method, collecting sufficient and accurate data and performing the actual valuation is put into action (Figure 9). The ESs are then translated to value through an economic value function or demand function (Ginsburg *et al.*, 2010). During this entire process, the various drawbacks and limitations of ES valuation techniques are considered and the results are statistically verified as far as possible (Ginsburg *et al.*, 2010).

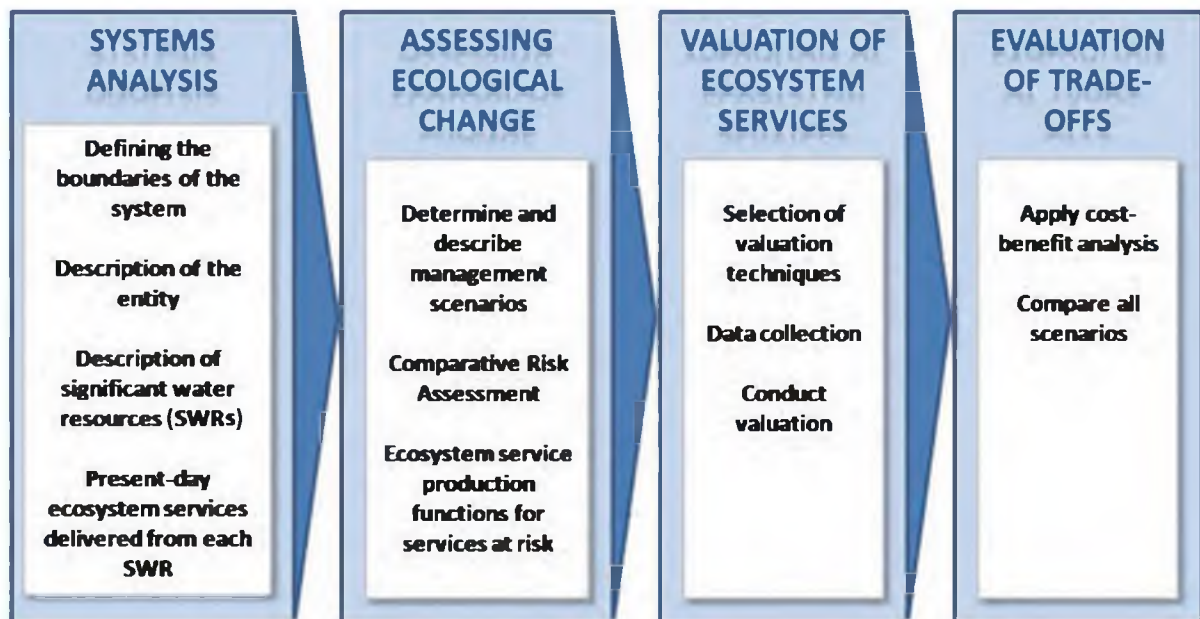


Figure 9: Simplified overview of the WRC aquatic ecosystem service evaluation framework (Adapted from Ginsburg *et al.*, 2010).

2.7.2. Challenges and Limitations of Ecosystem Service Valuation

A vast amount of literature exists that focuses on issues and limitations surrounding market and non-market valuation methods of ESs and nature in general. The limitations associated

with all types of ES valuation are primarily due to the fact that there are no formal markets for the majority of the goods and services and or markets are distorted in one way or another (Limburg, 2009; Pascual *et al.*, 2010; Turpie *et al.*, 2010; Van Den Berg *et al.*, 2013). Recently, there has been much debate around 'best methods' to use when valuing ESs, which is ultimately determined by the nature of the ESs themselves (Fisher *et al.*, 2008; Van Den Berg *et al.*, 2013). The inherent trade-offs between costs and benefits of ESs differ for all users as each have different requirements that make them challenging to value and manage (Raudsepp-Hearne *et al.*, 2010; Van Den Berg *et al.*, 2013).

The difficulty associated with quantifying and valuing ESs accordingly is due to a number of reasons. Information about specific goods or services is often flawed or non-existent (Limburg, 2009; Van Den Berg *et al.*, 2013). For example, benefits provided by aquatic ecosystems may not be accounted for in developing nations, where they are accounted for in developed nations. The markets for these AESs could distort their value due to external factors such as political decisions or media attention (Fisher *et al.*, 2008; Limburg, 2009). Furthermore, the complex nature of ESs cause them to be involved in multiple processes simultaneously, which makes it extremely difficult to analyse and compare multiple values at once (Costanza *et al.*, 2014; Limburg, 2009; Palmer *et al.*, 2014). There are further limitations and issues surrounding scale and uncertainty, ecosystems may be able to provide one type of service (at a particular quantity and quality) at a local scale but may contribute a different quantity and quality of services at larger scales (Brauman *et al.*, 2007; Pollard *et al.*, 2013; Van Wilgen *et al.*, 2008). The uncertainty associated with ecosystems speaks to their individual levels of resilience, which may vary drastically from place to place.

Monetary valuation of ESs, although not an end to itself, has been widely acknowledged as a powerful assessment and decision-making instrument providing a common metric from which comparisons can be drawn (Brauman *et al.*, 2007; Heal, 2000). Ultimately, basing policy and managerial action on models of human behaviour or value estimates of complex environmental systems is risky because many of these models are inappropriate for the contexts in which decisions are made (Hepburn *et al.*, 2010). Thus, this provides further incentive to investigate new methods to inform environmental, economic and social management and policy initiatives to understand the value of ESs without human agency being one of the dependent variables. Detailed discussions around the limitations, challenges

and criticisms of direct and indirect ES valuation methods are provided by Costanza *et al.* (2014), Ellis and Fisher (1987), Parks and Gowdy (2013), Pascual *et al.* (2010) and Sagoff (2011) amongst others.

2.8. Summary

Chapter 2 provides a detailed discussion around the concept of ESs, focusing on literature surrounding defining, classifying and identifying ESs. Relevant and topical tools for sustainable ES assessment and management were presented alongside decision-making frameworks. Defining final and intermediate AES was emphasised as this is one of the most important aspects of developing an AES based CLD. The concept of value in relation to ESs was briefly discussed using current schools of thought, the philosophy behind ES valuation was also presented. The South African approach towards evaluating AESs was outlined to demonstrate current practice. Lastly, general challenges and limitations associated with ES valuation methods were concisely discussed. Overall, this chapter provides a sound theoretical background behind the concept of ESs and develops a heuristic rationale for the study by emphasising theoretical and practical application of methods relevant to the research aim.

CHAPTER 3

SYSTEM DYNAMICS, CAUSAL LOOP DIAGRAMS AND VALUE CHAIN ANALYSES



Photo: J.M. Rawlins

3.1. Introduction

Nonlinearities and temporal and spatial lags are common features of natural systems, however, such system characteristics are often not considered in scientific and economic investigations (Costanza and Ruth, 1998). Consequently, this reduces the ability of these inquiries to provide accurate insights that are necessary to make appropriate management and policy decisions to improve resulting system behaviour and performance for complex socio-ecological systems (Lane, 2008). One division of systems thinking is called system dynamics and is designed to specifically address these system features and provide a holistic analysis of all system components that form a value chain for a specific good or service. A detailed understanding of system dynamics modelling, CLDs in particular, is vital to determine whether they can be used to conduct a value chain analysis. This chapter discusses system dynamics modelling with particular focus on the theory and application of CLDs and their limitations within the context of this study. A discourse on the notion of value chain analyses and associated limitations is presented in relation to ESs and the overall aim of the study.

3.2 System Dynamics

System dynamics is a problem evaluation approach based on the idea that the inherent structure of a system (the direction of causality or the way in which the key elements of the system are connected) generates and determines the systems behaviour (Kirkwood, 2013; Richardson and Pugh, 1989; Sterman, 2000). It is an interactive approach used to explain intricate ecosystem structure and function and illustrate the outcomes of potential management strategies by graphically representing system feedback structures. The system dynamics approach is used as a thinking model and simulation methodology that has been specifically developed to facilitate the study of dynamic behaviour in complex systems with special emphasis on the role of information feedback (Ford, 1999; Hjorth and Bagheri, 2006). Here the term 'complex' refers to the emergent properties of the ecosystem which cannot be described in terms of the separate components of the system, but rather in terms of the interactions between system components.

The original concepts and methodology were developed by Forrester (1961) and refined over the past five decades. Initially being applied for business and industrial system management, the scope and use of system dynamics has expanded to other focus areas including complex

socio-economic systems (Costanza and Ruth, 1998; Hjorth and Bagheri, 2006; Kelly, 1998; Kirkwood, 2013; Ruth, 1993). The term 'system' is used to describe an interdependent group of items or components forming a unified pattern (Kirkwood, 2013). A systems approach generally considers internal structure of a system as more important than external events regarding how a problem or change comes about (Kirkwood, 2013). System dynamics modelling is an approach that attempts to close temporal and spatial gaps in decision-making, actions and effects (Arquitt and Johnstone, 2008; Costanza and Ruth, 1998). Ultimately, system dynamics combines the theory, methods and philosophy required to analyse system behaviour drawing from an array of fields to understand system changes over time and to steer the system towards given objectives (Hjorth and Bagheri, 2006; Richardson, 1994).

System dynamics based studies begin with the conceptualisation of a problem and a set of assumptions that describe the nature of the concern (Forrester, 1961; Forrester, 1968). These assumptions are manifested in what is termed 'a mental model of a dynamic system' (MMODS), which provides information unattainable in any other manner (Lane, 2008). MMODSs have been referred to as accessible and enduring internal conceptual illustrations of extant, desired or planned systems whose structure preserves the perceived structure of that system (Doyle and Ford, 1998; Doyle and Ford, 2000; Lane, 1999; Lane, 2008). "In systems dynamics, the term 'mental model' includes our beliefs about the networks of causes and effects that describe how a system operates, along with the boundary of the model (which variables are included and which are excluded) and the time horizon we consider relevant – our framing or articulation of the problem" (Sterman, 2000: 16). There are a number of computer programming tools and languages (e.g. Vensim and STELLA) which assist decision-makers and scientists to focus the mental model they have of specific system phenomena, to elaborate and augment this model so it can yield the dynamic outcomes hidden in their assumptions and understanding of the system under consideration (Costanza and Ruth, 1998). System dynamics diagramming methods have changed significantly over time, or been altered in their use, however, there is little empirical proof of the efficacy of these different methods (Lane, 2008). Figure 10 illustrates three different viewpoints of cause and effect; system dynamics modelling considers all three viewpoints but is centred around on non-linear feedbacks.

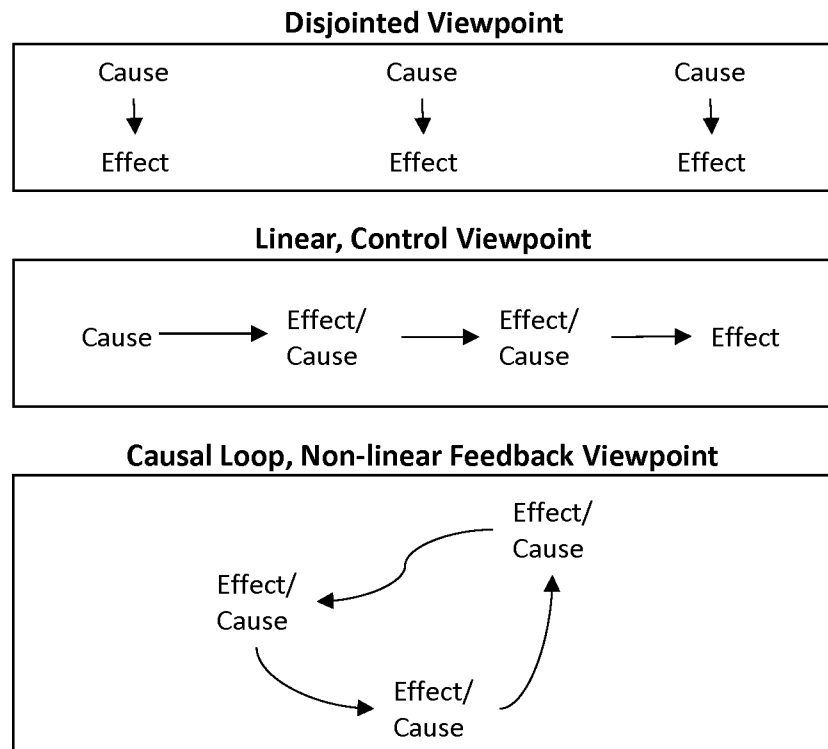


Figure 10: Three viewpoints of cause and effect (Adapted from Hitchins (2005)).

The field of system dynamics uses two overarching methods for solving complex problems; these are qualitative and quantitative modelling approaches (Hürlimann, 2009). This study focuses on qualitative modelling in the form of CLDs. Hürlimann (2009: 113-115) outlines the differences between qualitative and quantitative modelling in terms of model structure, variables, mode of operation, dynamics and interaction plus design. Both approaches have their own advantages and disadvantages; however, they are best used in conjunction with one another (Hürlimann, 2009; Landuyt *et al.*, 2013; Sterman, 2000; Wolstenholme, 1999). Although qualitative modelling is often used to evaluate policy options and business management, it has immense potential for illustrating and communicating complex ecological systems. This is because it allows for the communication of information about the structure of the system visually and without the use of technical jargon, while triggering discussion and debate among participants in order to create a common understanding regarding the system at hand (Stave, 2003; Vennix *et al.*, 1997).

Creating a system dynamics model not only creates buy-in from participating stakeholders, but also makes the underlying assumptions of the model explicit, which means the model can potentially be used to simulate the effect of ecosystem management actions and/or external

disturbances (scenarios) on the system as a whole. When system dynamics models are utilised interactively in a public forum they have the potential to illustrate resource systems and demonstrate different outcomes of strategies proposed by participants, managers and/or decision-makers (Stave, 2003; Vennix *et al.*, 1997).

Communication is the primary role of diagrams in system dynamics, they serve as “an intermediate transition between a verbal description and a set of equations” (Forrester, 1961: 81). Diagrams may be used for two types of communication known as ‘model conceptualisation’ and ‘model exposition’ (Lane, 2008). The former is when MMODSs are stimulated and framed in such a way that facilitates the use of system dynamics; diagrams are seen as evolving thinking tools that characterise an individual or group’s understanding of a particular issue and communicate their assumptions unambiguously, eventually leading to the creation of a fully formulated simulation model (Lane, 2008; Randers, 1980). The latter represents the assumptions that underpin mathematical models (Lane, 2008). Figure 11 uses a bathtub analogy to illustrate the different types of variables and causal links used in system dynamics.

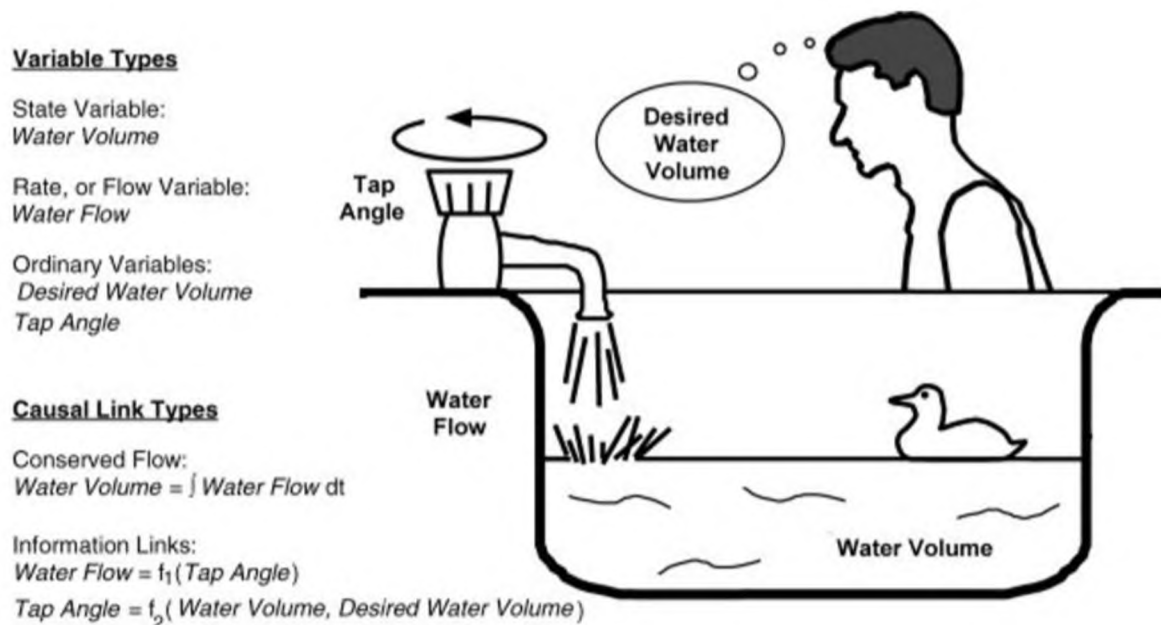


Figure 11: Illustration of variable types and causal links in system dynamics (Lane, 2008).

There are three types of variables and two types of causal links in system dynamics. ‘State variables’, ‘stocks’ or ‘levels’ (e.g. water volume) are only changed through

accumulation/drainage into/out of them while 'rate' or 'flow' variables (e.g. water flow) are the quantities or magnitude of those accumulating or draining processes (Lane, 2008). 'Ordinary' or 'auxiliary' (e.g. tap angle or desired water volume) variables do not directly influence stock variables. Causal links known as 'conserved flows' link flow and stock variables, while the values of auxiliary and flow variables are created immediately through 'information links' from other auxiliaries and state variables (Lane, 2008). The field of system dynamics allows for numerous diagramming approaches, however, the CLD and stock/flow diagram (SFD) are considered as the leading qualitative modelling diagramming conventions recognised by the global system dynamics community (Ford, 1999; Lane, 2008; Scholl, 1995). These two methods are relatively standardised and their benefits and shortfalls are generally well understood (Lane, 2008). Figure 12 illustrates the bathtub filling example (Figure 11) using a CLD and SFD, these examples highlight the differences between the two diagramming techniques and illustrate many of their advantages and disadvantages. The general structure, functioning, advantages and disadvantages of CLDs are discussed next. Ford (1999: 14-24), Sterman (2000: 191-262) and Lane (2008: 10-11) discuss these aspects of SFDs in more detail.

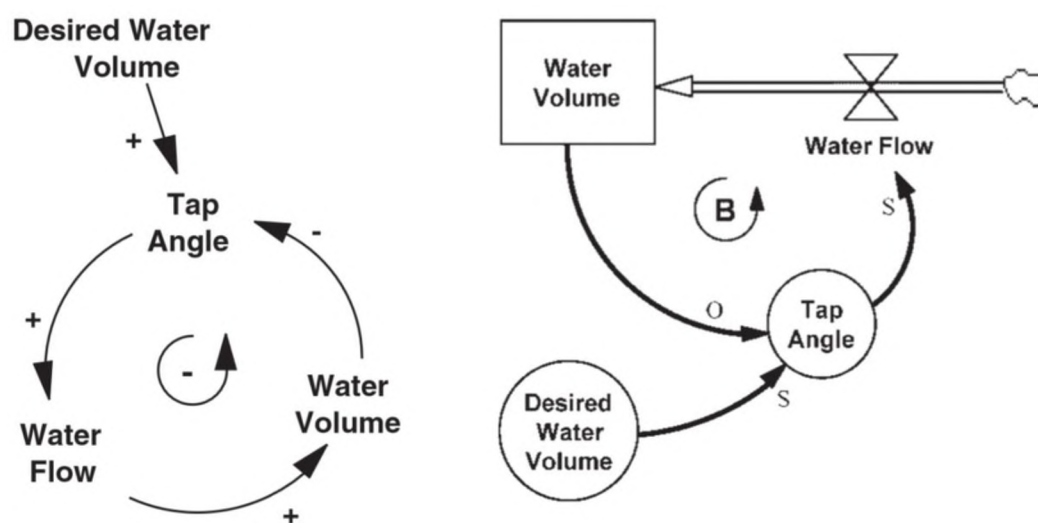


Figure 12: Causal loop diagram (left) and stock/flow diagram (right) illustrating the bathtub filling example explained in Figure 11 (Adapted from Lane (2008)).

A system dynamics analysis proceeds through several major steps: problem definition and delimiting, describing the underlying system, model development, model verification, modelling for analysis and communication (Ford, 1999; Richardson and Pugh, 1989). Participants and stakeholders in the modelling process can be involved at different stages of

the process. On the one extreme, it can be used simply for communicating the results of a completed model and/or to illustrate the effects of alternative options (Hale, 1993). However, a more participatory application will (where applicable) allow for the incorporation of the participant's suggestions and will enhance participant understanding of the resource system and ultimately improve consensus building and communication.

Costanza and Ruth (1998) illustrated three different cases where stakeholders were involved in the first of the above-mentioned steps (problem definition) for scoping resource management problems. Van den Belt (2000) emphasised the use of system dynamics for 'mediated modelling' within stakeholder groups, an example of involving stakeholders in the above-mentioned step 2 (system conceptualization). Vennix (1996) outlined several cases of stakeholder participation in model building (steps 1-4 above) and Guo *et al.* (2001) described the use of a system dynamics approach for general policy analysis (step 5) in environmental planning. These examples demonstrate the potential for each of these six steps to be applied to ES analysis within a sustainable management context.

3.2.1. Causal Loop Diagrams and Theory

CLDs, influence diagrams or cognitive maps are mostly considered the first stage of system dynamics modelling. The next stage would be to define the stocks and flows of the system and to quantify the interactions between elements and incorporate accompanying time delays (Ford, 1999; Sterman, 2000). CLDs are a qualitative diagramming language aimed at graphically illustrating feedback-driven systems (Kirkwood, 2013; Schaffernicht, 2010; Sterman, 2000).

Conceptualising a complex ecological system not only requires a clear definition of the key elements of the system and the cause and effect relationships between these elements, but also an account of the relationship of the system with other systems (Ford, 1999; Limburg *et al.*, 2002). Insight into context-relevant system interactions is therefore required to understand a complex ecosystem. The challenge to communicate the functionality of such a system lies within explaining the relationships between key elements of the system in a simple and transparent way. CLDs do exactly that, by facilitating a common understanding and improved insight among stakeholders *vis-à-vis* how the system works and why it responds to external stimuli the way it does (Evans, 2004; Ford, 1999; Richardson, 1997). Such insight

is immensely powerful for developing and strategising ecosystem management practices because it makes critical elements, which are within the stakeholder's control, explicit. This advances scenario building, scenario analysis and system projections; potentially allowing for the proactive formulation of ecosystem management strategies.

The reason for using CLDs to demonstrate natural systems stems from the issue that natural resource managers are often faced with, the challenge of building support and consensus for management strategies focussing on the trade-offs between environmental conservation and socio-economic benefits (Evans, 2004). These challenges are derived mainly from difficulty in communicating the value of ESs and the complexity of the underlying ecosystems to a broad audience consisting of different technical backgrounds and potentially conflicting perspectives (Costanza and Ruth, 1998; Lane, 2008; Morecroft, 1982; Sterman, 2000). Such support requires improved insight into the value of ESs through an improved understanding of the workings of the ecosystem itself.

A typical CLD comprises of a group of symbols representing a particular dynamic system's causal structure. This includes all relevant variables, causal links with a polarity (either negative or positive) and symbols which identify feedback loops and their polarity (Fernald *et al.*, 2012; Ford, 1999; Schaffernicht, 2010; Senge, 1990; Sterman, 2000). Each causal link has a direction and a polarity, delay marks are often included to provide an idea of a particular variable's behaviour over time (Ford, 1999; Richardson, 1986; Sterman, 2000; Schaffernicht, 2010). The polarity of conserved flows indicates whether the flow is draining out of or accumulating into a state variable while the relationship is purely functional between the stocks and or flows of information linkages (Lane, 2008). The time delay mark may be included because in some situations a rate of change may, during the early stages of influence on a particular system relationship, appear to be minor (i.e. the cause of change and the change itself may seem insignificant at first). However, as the change becomes magnified through reinforcing effects, the system impacts thereof can increase exponentially (Richardson, 1986; Schaffernicht, 2010; Sterman, 2000). It should be noted that all systems have some kind of built-in response delay, which can range from seconds to days, centuries or millions of years. It is critical to focus on one causal connection at a time when assigning + or – signs to causal linkages and all other variables need to be held constant (*ceteris paribus*) (Ford, 1999; Schaffernicht, 2010; Richardson, 1986; Sterman, 2000). This doctrine allows one to analyse the

impact of one change on the system at a time and increases the transparency of system analyses.

Each arrow is labelled with either a + or a – which represents the cause-and-effect relationship between the two variables (Ford, 1999). A + sign is used to represent a relationship where the two variables change in the same direction while a – sign indicates that the variables change in opposite directions (Ford, 1999; Richardson, 1986; Sterman, 2000). The arrows illustrating causal links always point from the independent variable to the dependent variable (Fernald *et al.*, 2012). When an element of a system indirectly influences itself, the portion of the system involved is called a feedback loop or a causal loop (Richardson and Pugh, 1981). The most relevant loops are highlighted by ‘a loop identifier’ which determines whether the loop is negative (balancing) or positive (reinforcing) (Sterman, 2000). Negative feedback loops act to negate outside disturbances while positive feedback loops act to amplify the impact of the disturbance (Ford, 1999; Lane, 2008). CLDs are primarily used to assist in communicating the structure of proposed conceptual models with the ultimate aim of improving complex system management and understanding (Fernald *et al.*, 2012; Ford, 1999; Kirkwood, 2013).

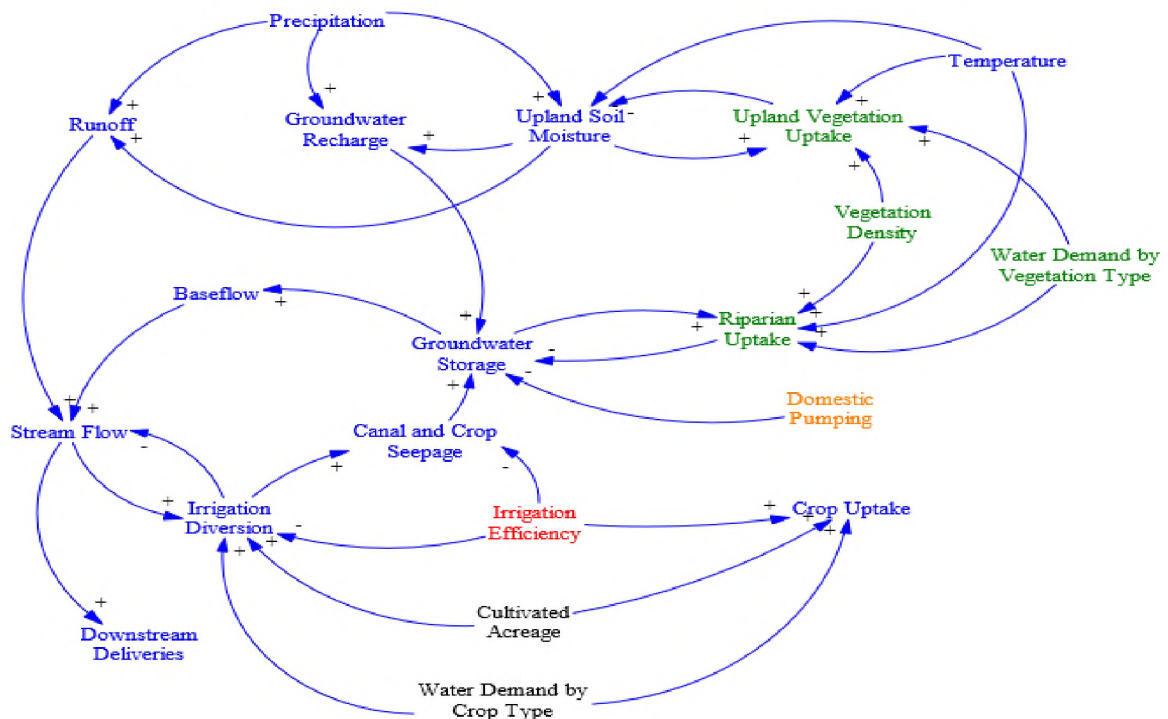


Figure 13: Hydrology subsystem causal loop diagram (Fernald et al., 2012).

Figure 13 is an example of a CLD that includes independent variables (e.g. runoff), dependent variables (e.g. stream flow) and arrows indicating causal links. Costanza and Ruth (1998), Evans (2004), Sterman (2000), Ford (1999) and Kirkwood (2013) elucidate how graphical descriptions of fundamental system components and their interactions are needed to better understand complex, nonlinear, dynamic systems. The different system components can be illustrated as a set of state variables, or stock variables (Costanza and Ruth, 1998). Stocks are variables in a system which exist at a point in time such as the amount of water in dam (Evans, 2004). These stocks are in turn influenced by controls, or flow variables (Costanza and Ruth, 1998). Flows are actions or activities that occur over a period of time and affect stocks in one way or another (defined by their polarity), such as the volume of water abstracted per month (Evans, 2004; Sterman, 2000). The extent of the variables (the magnitude of the flows) in turn may be dependent on the stocks themselves and other parameters that form part of the system (Costanza and Ruth, 1998). CLDs allow for stocks and flows to be 'created' to represent different elements in order to isolate particular effects, thus decision-makers and managers will be more informed as to what exactly causes specific outcomes in complex systems (Evans, 2004).

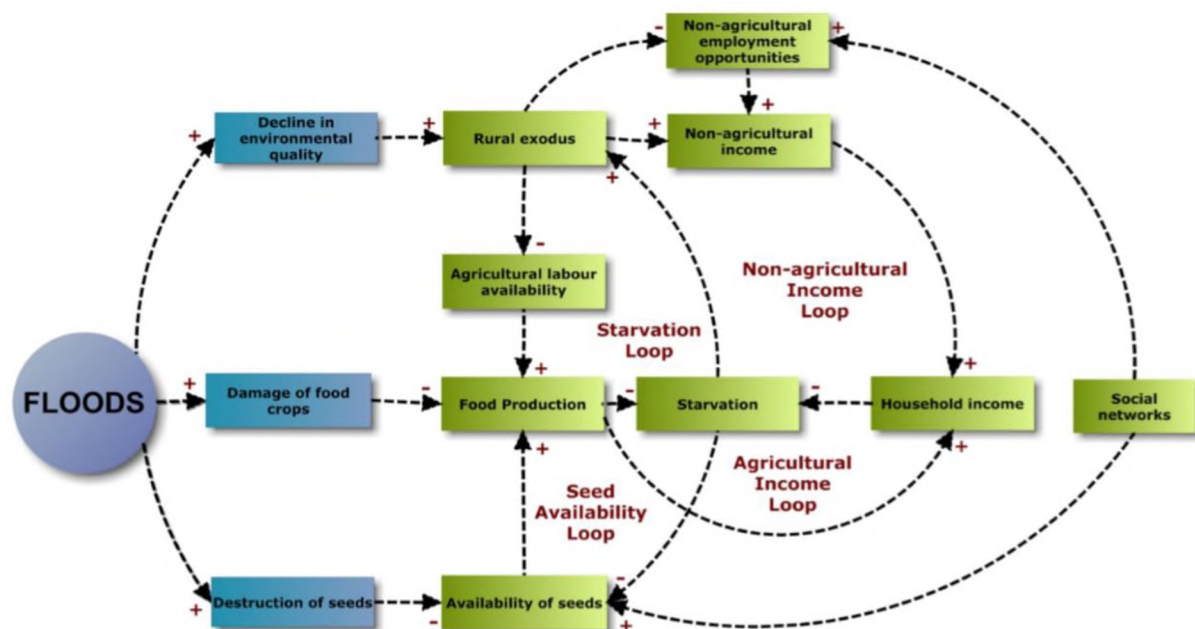


Figure 14: Simplified causal loop diagram illustrating the impact of floods on communities in Northern Ghana (Armah et al., 2010).

Armah *et al.* (2010) demonstrate an example of how qualitative modelling, CLDs specifically, can be used to conceptualise complex socio-economic systems and assist in the development of strategies to improve the management and decision-making within the system. The study explored the impact of flooding on rural communities in Northern Nigeria that are dependent on the supply of crucial ESs and natural resources. CLDs were used to develop and analyse flood-induced coping schemes, this enabled the identification of intrinsic characteristics of the socio-economic system under study that appeared to reduce vulnerability and mitigate flood damage risk (Armah *et al.*, 2010). Figure 14 demonstrates one of the CLDs that was used to analyse the system.

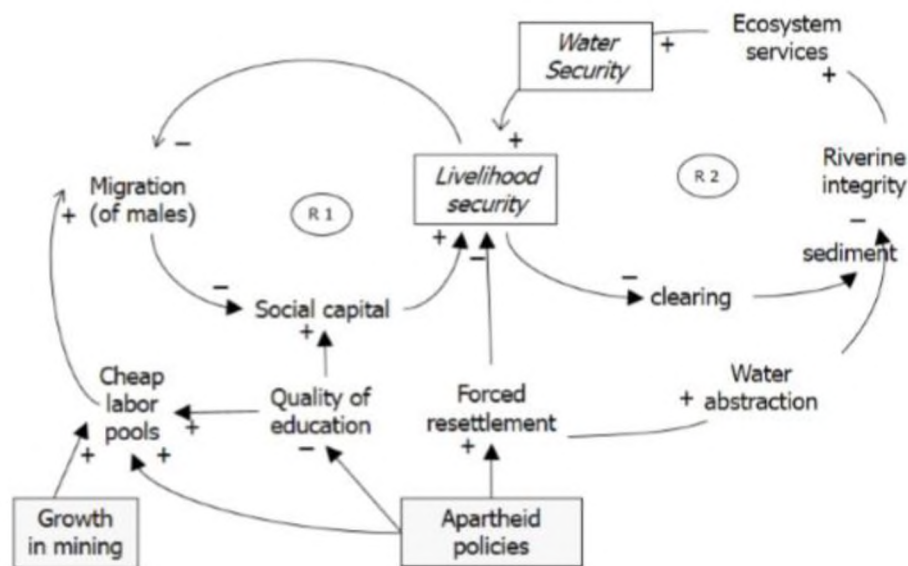


Figure 15: Causal loop diagram illustrating water security in the Sand River Catchment (Pollard *et al.*, 2014).

Pollard *et al.* (2008), Pollard *et al.* (2013) and Pollard *et al.* (2014) outline a number of scenarios where CLDs, or similar types of qualitative modelling tools, were used to map different components of a complex socio-economic system focusing on the Sand River Catchment. Pollard *et al.* (2014) demonstrate how a CLD can be used to provide a systemic view of water security, including the role of ESs (Figure 15). This process formed part of the integrative framework for context based decision-making in natural resource management outlined in section 2.5. However, ESs were considered as a whole as opposed to the individual services and their associated supporting functions and/or processes. During the development of a framework designed to assist with the understanding of AESs, concept maps were

developed to illustrate the ESs identified and other complex feedbacks within the system (Pollard *et al.*, 2013). However, the nature of different feedbacks were not defined in terms of causality, but these concept maps still remain an integral step towards developing more detailed CLDs.

Considering the discussion above it is evident that CLDs have the potential to be used as an alternative value chain analysis tool for ESs, which will explore the growth path beyond the subsistence level. The use of CLDs will allow for the identification of intermediate and final ESs in order to support the valuation process and improve system understanding, and ultimately, sustainable management.

3.2.2. Benefits and Shortfalls of Causal Loop Diagrams

CLDs have significant advantages and disadvantages for different applications, thus, it is imperative to identify these benefits and shortfalls to determine how they should be applied in the context of this project. Above-mentioned systems thinking typically requires a paradigm shift from examining individual system components to the examination of whole systems, including all underlying interacting components (Kirkwood, 2013; Sterman, 2000). CLDs facilitate this shift by making use of a diagram to present the system instead of cumbersome verbal descriptions of the relationships between elements (Morecroft, 1982). This is effective because verbal/spoken language presents interrelations through linear cause-and-effect chains (in order to speak within a 'rational' and understandable fashion), while system diagrams and depictions can reveal multiple and circular chains of cause-and-effect; which, in combination, explain the complexity of a system in a much clearer fashion (Costanza and Ruth, 1998; Evans, 2004; Kirkwood, 2013). Finnoff *et al.* (2005) describe the importance of analysing feedbacks in complex natural systems that interact with social and/or economic systems, as the consequences of ignoring such feedback could be devastating for either system.

Table 2 summarises the main strengths and weaknesses of CLDs for the purpose of system dynamics. It should be noted that this information is a result of the craft experience of numerous system dynamicists and numerous circumstances of using CLDs to apply system dynamics because quantitatively verified research in this regard is almost non-existent (Forrester, 1975; Lane, 2008; Morecroft and Sterman, 1994; Richardson, 1996). Lane (2008:

11-14) and Schaffernicht (2010: 665) describe the advantages and disadvantages of CLDs listed in Table 2 in more detail.

Table 2: Advantages and limitations of causal loop diagrams (Adapted from Lane (2008)).

Advantages	Limitations
- Focus is on Feedback - Interest in decision points and performance measures	Loss of distinctions between stock and flow variables, and conserved flows and information links can be unhelpful or misleading
Simplicity of language allows organic introduction into discussions	Can lead to mislabelling of loop polarities
Suppression of detail can be attractive to senior staff looking for overview	Can lack precision
Good rapid prototyping tool	System behaviour is only inferred
Only requires basic software or writing space	
Clearly communicates the location of causal loops	

Diagrams such as CLDs have proven to be effective in conveying quantitative and qualitative information of a complex nature (Larkin and Simon, 1987; Tufte, 1990). Ford (1999) and Sterman (2000) describe how CLDs are excellent for capturing hypotheses about the causes of dynamics and communicating important feedback responsible for problems in stochastic systems. CLDs have also been widely used for a number of different applications as they are capable of transparently including social, environmental and economic elements simultaneously (Costanza and Ruth, 1998; Evans, 2004; Kirkwood, 2013; Richardson, 1994). Hitchens (2005) discusses an advantage of using CLDs as forcing the user to tie in the ends of the different variables and their effects on one another, which assists in eliminating misconceptions. CLDs are highly effective at mapping the potential full effect (over space and time) of a particular project, program or policy (Evans, 2004; Morecroft, 1982; Sarriot *et al.*, 2015). With CLDs, practitioners no longer need to focus on one interaction between two variables, but can concentrate on the entire system, including the different variables and their many causes and effects. Britton *et al.* (2012) argue that CLDs allow one to create a holistic view of the system, identify cause-effect sequences and describe qualitative system properties which can provide essential information to managers and decision-makers. It is

evident that CLDs can help to better understand complex process and system dynamics, which is vital because system-modelling has grown considerably in recent years in many different spheres of influence (Senge and Sterman, 1992).

On the other hand, CLDs are limited in many aspects and should be thoroughly understood before being completely relied upon to understand complex, nonlinear system processes. One of the primary issues with CLDs is that they fail to make distinctions between information (proportional) links and rate-to-level (additive) links (also called 'conserved flows'), only illustrating the basic relationships often excludes the other minor relationships (Hürlimann, 2009; Richardson, 1986). Richardson (1997) refers to a key critique of CLDs concerning the different definitions of the term polarity. There are two main definitions of polarity, namely a 'complete' and a 'truncated' definition (Lane, 2008). Sterman (2000), Lane (2008) and Schaffernicht (2010) describe the 'complete' definition as: positive - when the independent variable changes with a positive sign, then the corresponding values of the dependent variable will be more (or less when the sign is negative) than it would have been otherwise. However, the basic or 'truncated' definition is described as: when a positive independent variable changes, then the dependent variable changes in the same direction (opposite direction when independent variable is negative) (Schaffernicht, 2010). Polarity and causal structure form part of the conceptual fundament of system dynamics; viewing CLDs as a language for conceptualising relevant aspects of causal structure has been debated in the literature for their apparent benefits and shortfalls (Lane, 2008; Richardson, 1997; Schaffernicht, 2010). Both definitions have their own advantages and disadvantages but generally the 'complete' definition refers to relative behaviour as opposed to events and can be difficult to conceptualise while the 'truncated' version often leads to faulty inferences (Schaffernicht, 2010).

CLDs, and system dynamics in general, have traditionally incorporated continuous behaviour as opposed to events; however, there is no proposed operational definition of the relationship between events and behaviour in terms of system dynamics (Richardson, 1986; Richardson, 1997; Schaffernicht, 2010). This could potentially obscure differences between approaches and is thus a fundamental limitation of CLDs. CLDs are not able to represent current levels of a particular variable's influence on another variable's behaviour or specifically how each influence functions (Kirkwood, 2013; Schaffernicht, 2010).

Consequently, the relationship between event and behaviour is regularly counterintuitive and if the definition of polarity facilitates faulty conclusions or is too complex to be properly used, then CLDs cannot be effective at holistically representing stochastic systems (Schaffernicht, 2010). Schaffernicht (2010: 665) uses an example of the general population model to outline a number of general but fundamental limitations of CLDs which coincide with those described by Lane (2008) (Table 2). CLDs can be a poor monitor of system behaviour if it contains several individual causal loops, because without simulation one must rely on intuition to infer the behaviour implied by the structure of a CLD (Forrester, 1994; Lane, 2008; Richmond, 1994).

The *ceteris paribus* condition (discussed in section 3.2.1), which is applied to CLDs and most qualitative modelling approaches, can be considered a shortfall as there are always more variables which are affected when changes are made to natural, stochastic systems (Ford, 1999; Richardson, 1997). This condition can potentially lead to faulty inferences about system behaviour. CLDs are not capable of accurately representing cause and effect over time (despite the inclusion of delay marks), which causes one to implicitly assume the effect of an intervention on a particular control variable will be realised immediately, this has been termed 'static thinking' (Moxnes, 2004; Schaffernicht, 2010). There are widespread concerns surrounding the effectiveness of CLDs to represent system changes at differing temporal and spatial scales. However, the scale of change/effect can be specified for individual models (Ford, 1999; Sterman, 2000). Issues surrounding the spatial dimension of CLDs can be accounted for to a certain degree through the incorporation of GIS analyses and maps that relate to the elements of the system. CLDs are able to display the nature of relationships between the different elements of a system but cannot describe the qualitative characteristics between these components (Richardson, 1994; Schaffernicht, 2010).

Ultimately it is difficult to conclude whether the advantages of using CLDs outweigh the disadvantages, or vice-versa, but rather the CLD approach is more applicable for certain applications than others. This serves as further motivation to determine the potential of CLDs to evaluate the value chains of AESs.

3.3. Value Chain Analysis

A value chain analysis (in the traditional sense) traces the value being added in each step in the life cycle of a particular good or service, from the process of production/harvesting right

through to final consumption or utilisation and waste disposal (Baleta and Pegram, 2014; Kaplinsky and Morris, 2000). Lanen *et al.* (2008) define a value chain as the group of activities that convert raw sources into the goods and services end users purchase and consume, and the treatment or disposal of any waste generated via the end user. While Mowen and Hansen (2011: 27) define a value chain as “a set of activities required to design, develop, produce, market, deliver and provide post-sales service for the product and services sold to the customer”. Value chain analyses are conceptual frameworks used to map and categorise chosen economic, social and environmental processes in service and product value chains, ultimately aiming to help create a better understanding of how and where enterprises and organisations are positioned within the value chain and identifying opportunities and potential leverage points for improvement (Sterman, 2000; Van Den Berg et al., 2013). Lynch (2003) reasons that the aim of a value chain is to maximise value creation through the entire chain while simultaneously minimising costs, where all company activities link together efficiently. Thus, when value chains are capable of creating additional value but with no additional private costs incurred, a competitive advantage is realised (Porter, 1985; Jordaan and Grové, 2008).

Kaplinsky (2000) emphasises that notable developments in value chain analysis frameworks are beginning to provide analytical structure that offers important insights into two primary concerns. These are the determinants of global income distribution and identification of constructive and efficacious policies to ameliorate trends towards unequalisation. Three important components of value chains need to be recognised in order to transform the approach from a heuristic tool into an analytical device (Kaplinsky, 2004). Firstly, value chains are repositories for dynamic rent, secondly, there is always some degree of governance in effectively functioning value chains and, lastly, ‘systematic-efficiency’ gives rise to effective value chains as opposed to ‘point-efficiency’ (Kaplinsky, 2004). Value chains are often used to investigate governance, particularly the relationships, interactions and power between different chain actors (Humphrey and Schmitz, 2001). The ‘governance’ concept is pivotal to global value chain approaches and has been used to analyse various institutional mechanisms and relationships through which non-market coordination of activities within a chain take place (Humphrey and Schmitz, 2001; Stamm and Von Drachenfels, 2011). It is evident from the variability in application processes that potential application opportunities of value chain

analyses are highly context specific, thus, it is imperative to use a holistic approach and focus on the objectives and desired outcomes of the chosen analysis.

According to USAID (2007), the structure of a value chain can be characterised in terms of five different elements. Firstly, the 'end market', which is the starting point of the analysis, refers to the people who consume or purchase the product or service (Jordaan and Grové, 2008). The second element being the 'business and enabling environment', which includes norms, laws (which incorporate biophysical laws of nature), customs, policies, regulations, public infrastructure, etc. that influence the way the product or service moves along the value chain to the end market. The third and fourth components involve vertical and horizontal linkages between various constituents within the value chain. Vertical linkages describe the relationship between the various actors at different levels along the value chain, while horizontal linkages describe the cooperation between different firms, which perform similar functions within the value chain itself (akin to causal system relationships). The fifth element involves the 'supporting markets' which include all support services such as financial services, legal advice, consulting services, underlying ecological functions, etc. that are key to firm-level and/or market-level improvements (Jordaan and Grové, 2008). However, as products and services become dematerialised and the value chains themselves no longer have physical dimensions, the concept of a value chain becomes an unsuitable tool to analyse many industries today (Peppard and Rylander, 2006). Thus, when considering the natural services provided by complex, non-linear ecosystems that have limited physical and financial dimensions within their value chains, it is clear that traditional, linear, firm-focused value chain analyses cannot be used to analyse these chains.

3.3.1. Value Chain Analysis Approaches

Recently, numerous different concepts have been used in other contexts that overlap with the concept of value chains, thus, there are a myriad of different approaches towards value chain analysis (Gereffi *et al.*, 2001; Kaplinsky and Morris, 2000). The most widely publicised concepts include Porter's value chain, supply chain, global commodity chain, French *filière*, global production network, actor-network theory, network value analysis and commodity chain (Jordaan and Grové, 2008; Law, 1999; Peppard and Rylander, 2006). The general

underlying concepts of each of these approaches are very similar; however, each have particular characteristics and nuances that distinguish them from one another.

Michael Porter introduced the term ‘value chain’ in his book *“Competitive Advantage – Creating and Sustaining Superior Performance”* (Porter, 1985). The value chain analysis is based on Porter’s generic value chain model, which was developed with the aim to explore Porter’s model of competitive advantage through cost leadership strategies and differentiation (Baum, 2013; Porter, 1985).

Ketchen and Hult (2007) describe Porter’s value chain alongside the process of disaggregating businesses into relevant strategic activities in order to understand the behaviour of costs and the current and potential future sources of differentiation (Porter, 1985). Hergert and Morris (1989: 183) state “the fundamental notion in the value chain analysis is that a product gains value as it passes through the vertical stream of production within the firm. When created value exceeds costs a profit is generated”. Even though the model was originally designed for companies in the manufacturing industry, it is being applied to increasingly more applications around the world (Ketchen and Hult 2007). Porter’s competitive advantage model will not be discussed as it is outside the scope of this study. Figure 16 illustrates Porter’s classical value chain of the firm which is separated into primary and support activities (Porter, 1985). Porter’s value chain is a basic analysis tool used to systematically explore all the activities performed within a firm as well as the interactions between the activities (Romero and Tejada, 2011).



Figure 16: Porter’s classical value chain of the firm (Porter, 1985).

Primary activities involve a product's physical creation, distribution and sales, and after-sales service (Mowen and Hansen, 2011); these are identified as value-adding activities (Lanen *et al.*, 2008). Support activities are designed to provide the assistance necessary for primary activities, they do not form part of the closer value chain but are included in every function of the value chain (Ireland *et al.*, 2009; Lanen *et al.*, 2008). Porter (1985) also introduced the concept of a value system which describes the system of value chains within which a firm operates that all contribute to moving the physical product towards the end buyer, these systems can also extend beyond the boundaries of an enterprise (Jordaan and Grové, 2008).

The approach to supply chain analyses has been used as a substitute to that of value chain analyses within similar contexts. Although, value chains focus downstream on adding value in the eyes of the customer/consumer while supply chains focus upstream on integrating suppliers' and producers' processes as well as on reducing waste (Feller *et al.*, 2006). Gereffi *et al.* (2001) described the term supply chain as a generic label for an input-output based structure of value adding activities, which begin with raw materials and end with the final product. Supply chains allow for a convenient description of the flow of the physical products from the input suppliers to the end consumer (Jordaan and Grové, 2008). Gereffi *et al.* (2001), Henderson *et al.* (2002), Jordaan and Grové (2008), Law (1999), Murdoch (1998), Peppard and Rylander (2006), Raikes *et al.* (2000), Stamm (2004), Sturgeon (2001) and Tallec and Bockel (2005) illustrate and discuss the most widely publicised and utilised value chain analysis approaches/concepts.

3.3.2. Issues and Limitations Surrounding Value Chain Analysis

Focusing on real world value chains, products move through corresponding links that may form part of alternate value chains, thus, these chains are generally much more complex than the definition of a simple value chain (Kaplinsky and Morris, 2000). Traditionally value chain analyses are geared towards linear processes and private goods that form part of a conventional neo-classical market system. Hence, the notion of incorporating public goods (such as ESs) which generally do not have defined market values nor are traded in formal markets, into a value chain analysis will require an alternative approach to the conventional linear techniques (Henderson *et al.*, 2002; Van Den Berg *et al.*, 2013).

ESs represent and carry with them multiple associations of value which are often explicitly economic but also social, cultural and environmental (Van Den Berg *et al.*, 2013). However, a multitude of analysis and valuation techniques exist that attempt to identify and present tradable value attributes of ESs but only allow for limited value chain analysis and market development processes (Brauman *et al.*, 2007).

The different approaches towards value chain analysis have been developed in response to a variety of challenges associated with individual methods and applications. Within value chain analyses exist a proliferation of overlapping concepts and terms that make it difficult to select the correct tool for a specific purpose (Gereffi *et al.*, 2001). Many of these confounding terms such as 'global commodity chains', 'value chains', 'value systems', 'production networks', and 'value networks' have common ground much greater than their distinguishing features. Ramsay (2005) and Jordaan and Grové (2008) highlight how the concept of a value chain could be considered to be a misnomer since only physical resources are transferred along the chain of linkages between actors/firms, effectively supplies going in one direction and money in the other. On the other hand, the term value is a 'meta-physical' perceived quality associated with the benefits that occur at the different points of exchange along the value chain (Jordaan and Grové, 2008).

Recently there has been a significant increase in the amount of value chain analysis research and case studies, however, many of the concepts surrounding the approach are still not very clear (Gereffi *et al.*, 2001). Without a well-defined and understood theoretical framework guiding value chain analyses, there are severe limits on both the outcomes that can be derived from diverse case studies and comparisons between different value chains and their subsequent analysis outcomes (Gereffi *et al.*, 2001). These approaches can be made more efficient and effective by developing shared parameters for defining different types of value chains and a clear taxonomy of value chain analyses that has the potential to be operationalised through the lens of a robust suite of indicators (Gereffi *et al.*, 2001). The intricate realities of the 'network economy' and complex environments within which we exist require that we reconsider traditional methods for analysing complex and competitive environments (Peppard and Rylander, 2006). It is evident that different methods of value chain analysis are only suitable for specific approaches, further emphasising the importance

of understanding the strengths and limitations of utilising CLDs as a tool for value chain analysis of ESs.

3.3.3. Ecosystem Services and Value Chains

The complex and dynamic nature of socio-ecological systems make system behaviour as a function of human and natural disturbances difficult to predict, nevertheless, there have been substantial improvements in the understanding of these systems (EC, 2013). Incorporating ESs thinking into value chain assessments is a relatively recent consideration and thus the literature is scarce (Van Den Berg *et al.*, 2013). Many ESs have been directly or indirectly addressed through approaches to increase the sustainability of value chains, these include certification schemes, corporate social responsibility, risk management and mitigation initiatives (Grigg *et al.*, 2009; Weiss *et al.*, 2011). Christensen *et al.* (2011) demonstrated a combined economic and ecological approach, aimed at giving equal emphasis to both disciplines. The study focused on integrating the tropic ecosystem model into a value chain approach to track the flows (i.e. amounts, costs and revenue) of fish products from the sea through to the final consumer in order to assess the social and economic features of fish trade and production (Christensen *et al.*, 2011).

There have been numerous multi-actor activities addressing how biodiversity is and can be integrated into value chains (Bolwig *et al.*, 2010; Van Den Berg *et al.*, 2013). Some of these include the IUCN Global Business and Biodiversity Program (BBP) (Bishop *et al.*, 2008), the EU Business and Biodiversity platform, the UNDP protecting biodiversity in working with agribusiness project (Leibel, 2012) and the Business and Biodiversity Offsets Program (BBOP) (Van Den Berg *et al.*, 2013). These initiatives and research endeavours emphasise the limits of market-based approaches for value chains, which range from unorganised and powerless workers and the lack of true market values for ESs to difficulties in product and service commercialisation (Brauman *et al.*, 2007; Van Den Berg *et al.*, 2013; Wood, 2001).

Van Den Berg *et al.* (2013) described a case study that explores the governance options available to the Dutch government for the promotion of the sustainable use of ESs within selected tropical timber value chains and how ESs can be given a more explicit role in the public and market mechanisms that govern tropical timber value chains. The study concludes that in most circumstances the term ESs has not yet been accurately and clearly defined,

which is argued to be a result of their strong links to markets and various attempts to define them in terms of economic value (Van Den Berg *et al.*, 2013). This further corroborates the need to develop a suitable tool or framework that can enable a value chain analysis for a wide range of ESs.

Bolwig *et al.* (2010) introduced a conceptual framework aimed at overcoming many limitations of traditional 'standalone' value chains through integrating 'vertical' and 'horizontal' components that affect poverty and sustainability. The conceptual framework considers local and global environmental concerns but does not address ESs specifically. It focuses on the nature and function of various actors within the value chain, how these affect the internal structure and composition of livelihoods and environmental concerns within the broader political economy as well as the transnational linkages and networks that exist along the value chain (Bolwig *et al.*, 2010). The dynamics of this particular framework and the nature in which it approaches value chain analysis can be loosely compared to the process and function of CLDs, as positive and negative impacts are represented in an attempt to simulate real-world complexity (specifically in terms of how local communities and the environment are affected) (Bolwig *et al.*, 2010). However, the fundamental difference is that the framework is geared towards analysing the actions of specific human actors within the value chain, taking a largely anthropocentric standpoint.

The notion of incorporating systems thinking and complexity theory into value chain analysis links directly with the concept of ESs, which are the products of highly complex and dynamic systems (Brauman *et al.*, 2007; Limburg, 2009). Hergert and Morris (1989: 183) describe the key concept in value chain analyses as being that a particular product or service will increase in value as it moves through various processes required to develop it into a final service. On the surface, this statement seems to hold true for complex ESs, as there are numerous underlying ecological processes and functions which increase the value of these goods and services over time until it is consumed and/or utilised (Fisher *et al.*, 2009; Johnston and Russell, 2011; Landers and Nahlik, 2013).

Stamm and Von Drachenfels (2011) emphasise how value chain analyses generally begin with linear mapping and categorising of elements/activities in the chain, from initial inputs all the way through to the final utilisation of the good or service. However, it is agreed that linear

understandings of value chains tend to be misleading as the ‘true value’ of goods and services can be found in more complex networks (Henderson *et al.*, 2002). New schools of thought are emerging around the idea that a value chain will have impacts on and will be impacted on by other peripheral value chains of associated goods and services, be it directly or indirectly. Value chain analyses have been used to address the issue of the conservation finance gap, although these analyses are done specifically to conserve ecosystems and are based on business’ value chains (Credit Suisse *et al.*, 2014). It is clear that the factors limiting value chain analyses of ESs are related to both the application of value chain analyses in different contexts and the complexity and variety of ESs, as well as the lack of accepted and standardised definitions.

3.3. Summary

Chapter 3 presents a comprehensive discussion around the topics of system dynamics, CLDs and value chain analyses. Particular focus was put on the theoretical and practical application of CLDs as well as the analysis of examples where they have been used in the environmental field, as this makes up a large component of this study. The concept of value chain analysis was presented and the background of their development discussed in relation to different objectives under specific decision contexts. Then the limitations and issues surrounding value chain analyses were put forward with particular respect to methodological challenges. Lastly, current literature connecting value chain analyses and the concept of ESs were presented and discussed in terms of the potential to conduct a value chain analysis for ESs. Ultimately, this chapter presents a rigorous theoretical background of CLDs and value chain analyses, two of the most critical components to the aim of this study. The discussion attempts to further develop the line of investigation by highlighting relevant information gaps in available literature.

CHAPTER 4

THE BAVIAANSKLOOF CATCHMENT



Photo: J.M. Rawlins

4.1 Rationale for Study Area Selection

The Baviaanskloof catchment was selected as an appropriate study area because there is already a large amount of scientific research and data available directly related to ESs (Boshoff, 2005; Glenday, pers. comm. 2015; Jansen, 2008; Mander *et al.*, 2010). The large amount of environmental, ecological and economic research focusing on the Baviaanskloof means that a significant number of experts in these fields have first-hand experience in the study area. This is beneficial for hosting specialised workshops, as there is a relatively large base of professionals to approach. There are several different user groups, primarily irrigated agriculture, livestock and game farming, conservation and recreation/tourism, which compete for these AESs at different levels (Boshoff *et al.*, 2000; Illgner and Haigh, 2003; Nel *et al.*, 2006). Moreover, numerous civil society organisations such as Working for Water (WfW) and the Subtropical Thicket Restoration Programme (STRP) are involved with different ecosystem restoration projects throughout the catchment (Boshoff, 2005; Nel *et al.*, 2006). Thus, trade-offs between different ESs for diverse user groups can be considered during the value chain analysis.

Nel *et al.* (2006) note how the Baviaanskloof is a priority biodiversity conservation area for rivers. Additionally, the Baviaanskloof is considered part of the Kougaberg Strategic Water Source Area (SWSA). SWSAs are areas that supply a disproportionate amount of mean annual runoff to a geographical region of interest (Nel *et al.*, 2013; Nel *et al.*, 2011; SANBI, 2007). SWSAs have the potential to contribute significantly to overall water quality and supply, supporting development needs that are often a large distance away (Nel *et al.*, 2013). Thus, research aimed at developing tools to improve aquatic ecosystem functioning and service delivery will be most beneficial focusing on South Africa's SWSAs.

The climatic and topographical variations (explained in section 4.2) that exist in the Baviaanskloof River ecosystems facilitate a large number of different ESs, as well as that the majority of the land cover is natural veld (Kruger, pers. Comm. 2015; Mendelsohn, pers. comm. 2015; Skowno, 2008). The extent of the study area is large enough to consider a diverse spread of AESs while still conforming to the strict ES identification and categorisation requirements outlined in sections 2.2, 2.3 and 2.4. The default ecological management class assigned to the Baviaanskloof is category B which means that the ecological condition of the

area is considered to be 'largely natural' (DWAF, 2002; Nel *et al.*, 2011). Due to the broad and novel nature of the approach being applied in this project, considering an area that is 'largely natural' (minimal anthropogenic disturbances) will simplify the process of identifying appropriate and relevant AESs and describing their complex interactions. The impact of chosen scenarios on the provision of final AESs will be easier to predict in a system with minimal anthropogenic disturbances (Walker *et al.*, 2002).

4.2. Physical Attributes of the Study Area

4.2.1 Location and Topography

Figure 17a illustrates the location of the Baviaanskloof catchment (approximate coordinates range from 23°35'E to 24°25'E and 33°30'S to 33°45'S) within South Africa, the catchment boundaries physically delineate the study area. The primary catchment is divided into four quaternary catchments (L81 A – D), which are the principal water management units in South Africa (illustrated in Figure 17c). This area is known as the 'true' Baviaanskloof, as this is the catchment through which the Baviaanskloof River runs, despite the Baviaanskloof Mega Reserve (BMR) extending into several surrounding catchment areas as well (e.g. Kouga and Gamtoos) (Boshoff, 2005; DWAF, 2002; Jansen, 2008).

The Cape Fold Mountains account for the rugged terrain that features over the majority of the catchment apart from the valley floors which are generally flat floodplains typical of highly eroded valleys (DWAF, 2003; Jansen, 2008; Knight, 2012). The mountains are generally very steep, with only one third of the slopes exhibiting a gradient of less than 30% (Illgner and Haigh, 2003). The altitude of the mountains north and south of the Baviaanskloof River reach heights of up to 1758 meters above sea level (masl) while the valley floor elevation ranges from approximately 700 to 300 masl (Jansen, 2008; Mander *et al.*, 2010). South of the river is the Kouga mountain range and to the north is the Baviaanskloof mountain range, these are evident from the digital elevation model (DEM) shown in Figure 17b. Over time, the force of the river eroded the valley between the two mountain ranges which divides the floodplain into individual alluvial plains as a result of higher-lying erosion-resistant rock (DWAF, 2003; Mander *et al.*, 2010). Naturally, the lowest point of the river system (lower than 160 masl) exists at the confluence with the Kouga River at the far east of the catchment (Illgner and Haigh, 2003; Jansen, 2008).

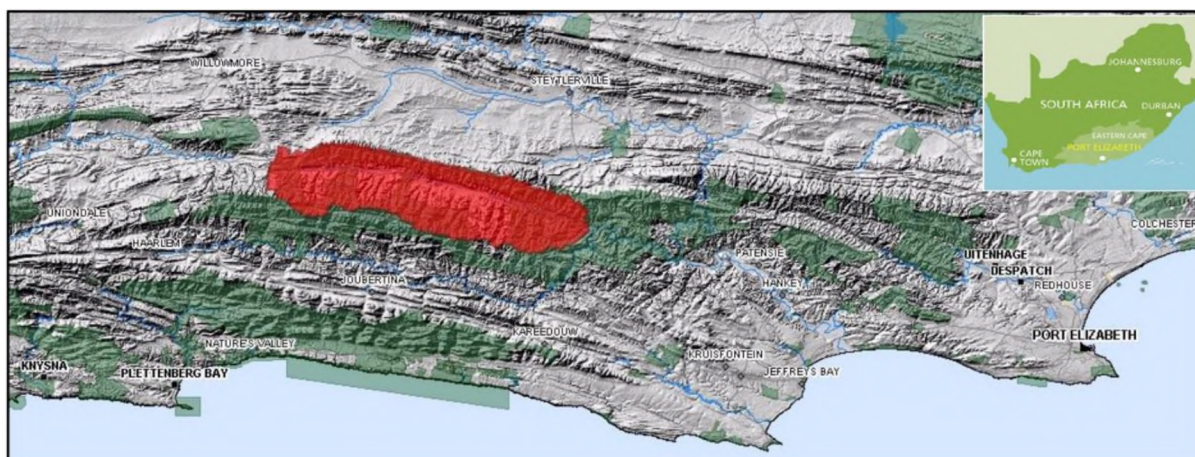


Figure 17a: Location of the Baviaanskloof Catchment (Adapted from Knight (2012)).

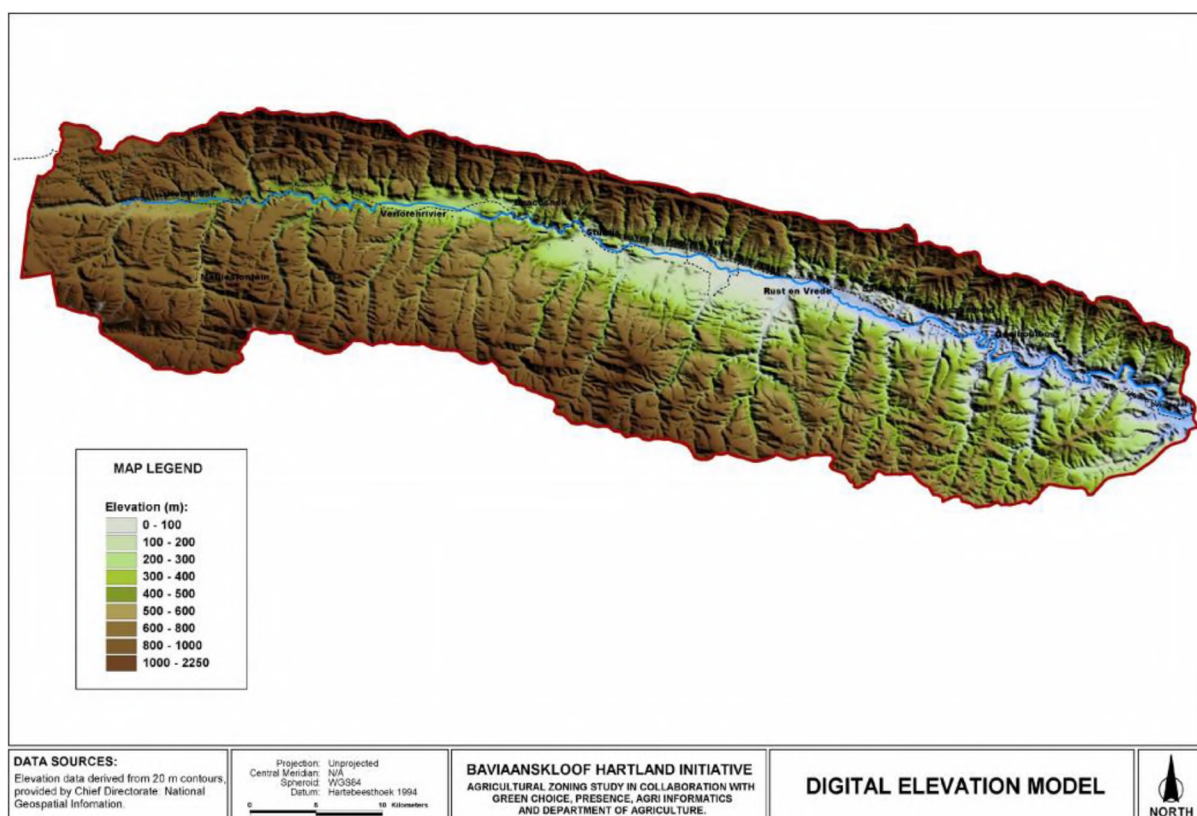


Figure 17b: Digital Elevation Model of the Baviaanskloof Catchment (Knight, 2012).

4.2.2 Climate and Hydrology

The Baviaanskloof is situated in a bimodal rainfall zone with spring and autumn maxima, with an annual average of approximately 350 mm and a large interannual variability (ranging from less than 100 mm to in excess of 700 mm) (Mander *et al.*, 2010). Figure 17c illustrates the distribution of mean annual precipitation (MAP) and mean annual evaporation (MAE) over

the catchment. It is evident from the diagram that the south-west section of the catchment receives the highest rainfall while most of the catchment receives between 301-400 mm per annum. The rain is primarily convective and/or orographic in nature with more than two thirds falling in the summer months (Jansen, 2008). The MAP is characteristically 20% of the MAE subsequently resulting in arid conditions through most of the catchment (Illgner and Haigh, 2003; Van Luijk *et al.*, 2013). Ephemeral streams dominate the landscape due to the high infiltration and evaporation rates coupled with steep topography and short but intense rainfall events (Illgner and Haigh, 2003; Knight, 2012; Van Der Burg, 2008).

Following a similar trend to the rainfall, average temperatures exhibit high interannual variability ranging anywhere from 32 °C to -3 °C (Boshoff, 2005; Mander *et al.*, 2010). Frost occurs frequently during the winter months as well as snow on many of the mountaintops. During winter months, the prevailing wind direction is northwest and southeast during summer (Jansen, 2008). The unpredictable climatic conditions and arid conditions emphasise the importance of reliable AES provision for the beneficiaries within and downstream of the catchment.

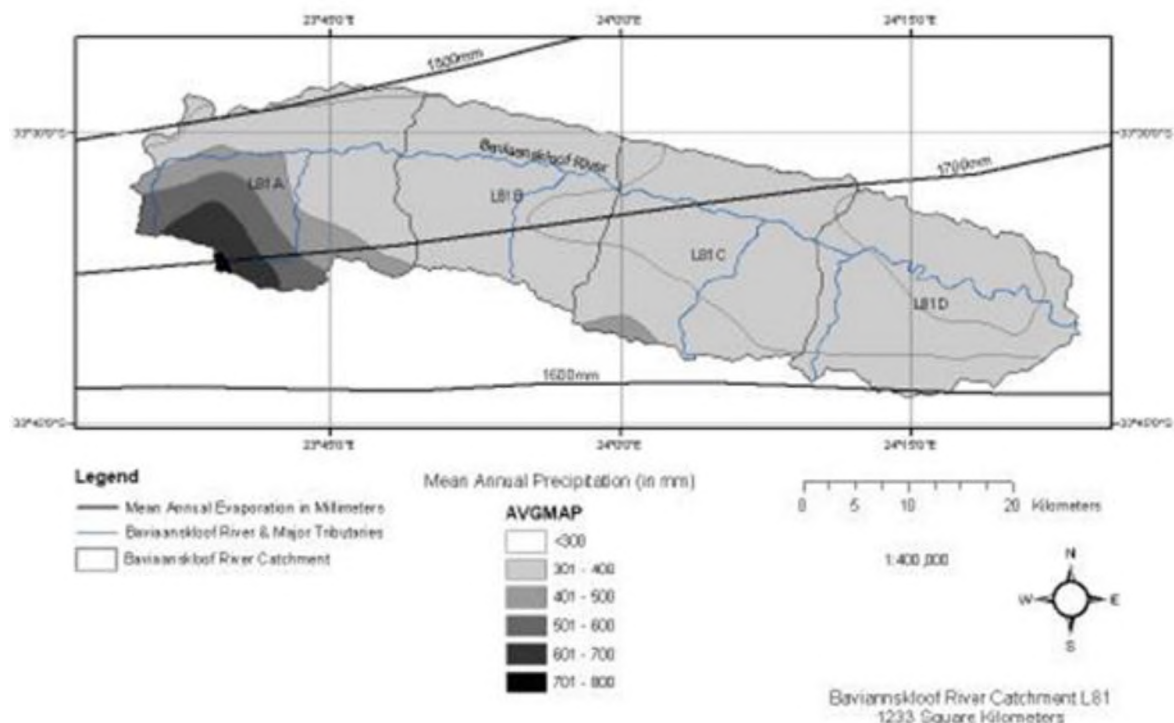


Figure 17c: Map of the Baviaanskloof Quaternary Catchments Showing the Prevailing MAP and MAE (Illgner and Haigh, 2003).

4.2.3 Geology

Quartzites and sandstones of the Table Mountain Group (TMG) make up the dominant geological formation of the Baviaanskloof with small amounts of shale that originate from the Bokkeveld group, both of which have been subject to intensive folding (Illgner and Haigh, 2003; Rust and Illenberger, 1989; Welman and Barnard, 2007). These two formations are overlain by intermittent outcrops of the Enon Conglomerate, which contains alluvial gravels and sands consisting of angular to rounded pebbles in a sandy matrix (DWAF, 2003; Illgner and Haigh, 2003; Jansen, 2008; Welman and Barnard, 2007). The arenaceous rocks of the TMG make up the vast majority of the Kouga and Baviaanskloof mountains while the valley floors consist mainly of shales from the Bokkeveld Group (Welman and Barnard, 2007). Stratified sandy, alluvial soils originating from the fine grained, sedimentary nature of the Bokkeveld Group are dominant in the valley, exhibiting high porosity and drainage (DWAF, 2003; Welman and Barnard, 2007).

Welman and Barnard (2007) describe the hydrogeology of the Baviaanskloof primarily through distinguishing between two groundwater systems, the 'TMG fractured aquifer' and the 'conglomerate/alluvial aquifer'. The TMG aquifer is an interspersed network of fractures and joints that facilitate groundwater movement; however, various isolated systems remain where there are no connections (Jansen, 2008). This system is largely situated within and below the two mountain ranges, these being relatively high rainfall areas support the groundwater recharge (Welman and Barnard, 2007; Yong, 2005). The alluvial aquifer is considered as a primary aquifer that runs the length of the Baviaanskloof River and is made of Enon Conglomerate with cobbles, pebbles and clay lenses (DWAF, 2003; Welman and Barnard, 2007). The aquifer makes direct contact with the Baviaanskloof River and recharges via runoff from mountain slopes and rainfall in the valley; this allows the aquifer to maintain the base flow of the river during dry periods (Jansen, 2008; Welman and Barnard, 2007). Much of the surrounding agriculture uses water from the alluvial aquifer for irrigation purposes. Jansen (2008) describes that there is no evidence of whether the two different aquifer systems are directly connected or not. Understanding the role that geology plays in the delivery of AESs is imperative for the purpose of this research.

4.2.4 Surface Water and Surrounding Catchments

Figure 17d illustrates the delineation of and interrelations between the Gamtoos, Groot, Kouga and Baviaanskloof catchments, the hydrology of one catchment is directly influenced by those adjacent. The Kouga and Baviaanskloof Rivers flow into their respective catchments and meet at the Kouga Dam, from there the water is released into the Gamtoos catchment. Water flows from the Groot catchment into the Gamtoos catchment and small amounts also flow peripherally into the Baviaanskloof catchment. However, due to excess mineralization, the water from the Groot catchment is hydrologically separated from the water in the Kouga Dam (DWAF, 2002; Jansen, 2008).

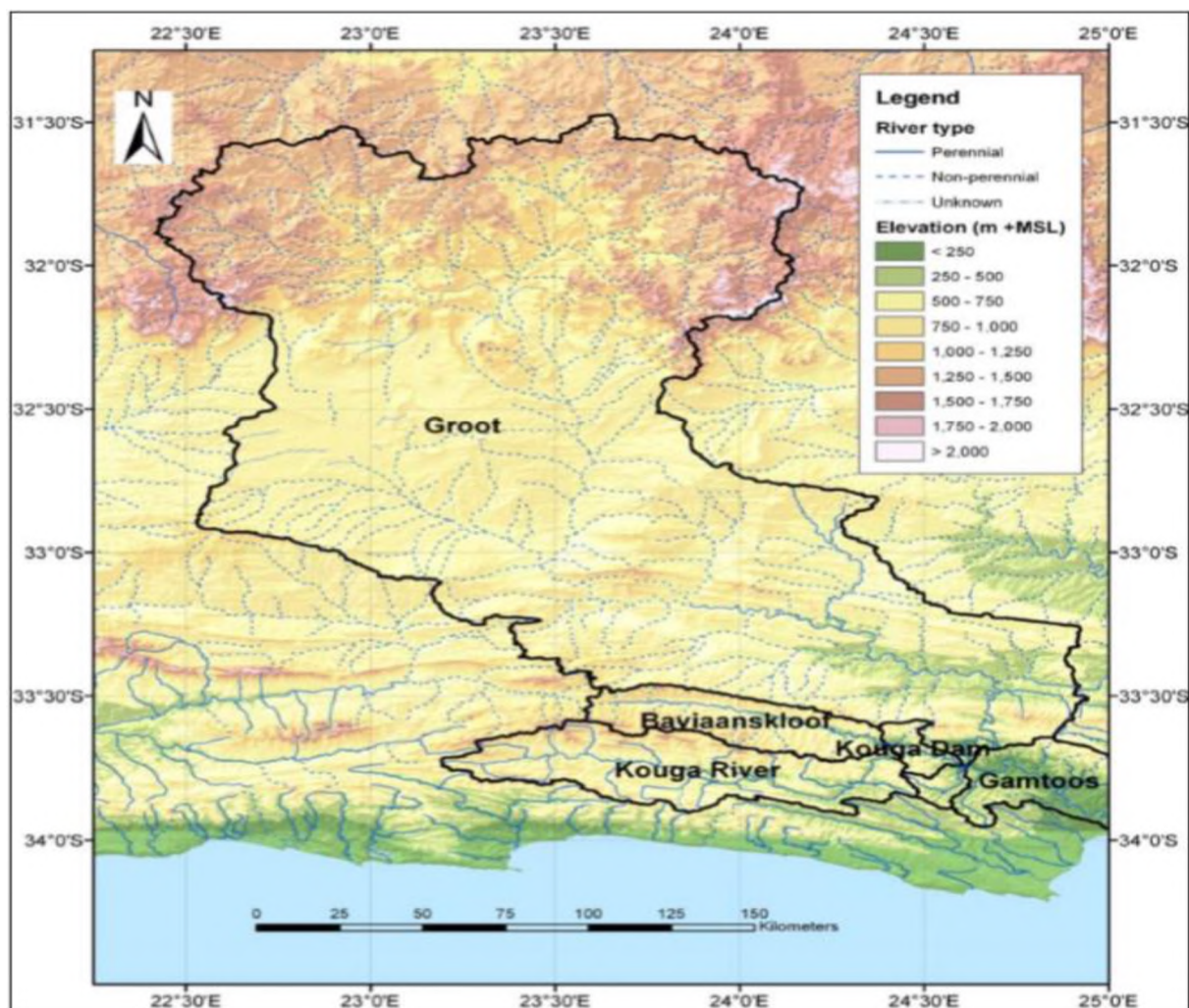


Figure 17d: Catchments Surrounding the Baviaanskloof (Adapted from Jansen (2008)).

The Baviaanskloof River is not always visible throughout the year as many river sections continue as ephemeral interflow through permeable sections of the alluvial deposits (Jansen,

2008). The Baviaanskloof is a perennial river with a number of non-perennial tributaries that flow down the valley into the river, many of which are canalised to prevent flooding on cultivated land (Jansen, 2008). The Baviaanskloof River is generally characterised by clear waters and low conductivity (DWAF, 2002). Despite the study area being restricted explicitly to the Baviaanskloof catchment, many AESs benefit people outside of the catchment and numerous processes external to the catchment can affect the provision of these services (Mander *et al.*, 2010). Hence, it is critical to understand the hydrological system as a whole and then narrow the focus down to the catchment under study.

4.2.5 Vegetation

The Baviaanskloof falls within the eastern end of the Cape Floristic Region (CFR) and is thus part of the smallest and most distinctive of the six plant kingdoms (Boshoff, 2005; DWAF, 2003; Knight, 2012; Noirtin, 2008; Working for Wetlands Programme, 2015). The Grassland, Subtropical Thicket, Nama-karoo and Fynbos biomes occur marginally in the Baviaanskloof catchment while seven of South Africa's eight biomes are considered to occur in the broader Baviaanskloof (Boshoff, 2005; Kirkman, 2006; Proches and Cowling, 2007). Fynbos and subtropical thicket species dominate the CFR (Boshoff, 2005). Euston-Brown (2006) and Mander *et al.* (2010) further describe the complex vegetation community, structure and composition of the Baviaanskloof as being dominated by Fynbos and biome-transitional vegetation types. Vegetation type, structure and function play an important role in the generation and regulation of AESs (Mander *et al.*, 2010; Noirtin, 2008; Powell, 2009).

4.3 Land Use

The largest portion of land (approximately 65%) in the Baviaanskloof is used as a nature area or conservancy (BMR) which is used for watershed and biodiversity management, while the rest of the land is used for agriculture, game farming, settlement and tourism (Boshoff, 2005; DWAF, 2002; Kirkman, 2006; Knight, 2012). Farm sizes vary significantly and most agricultural land is used for livestock farming, primarily goats and sheep. Relative to this, a small portion of the land is used for irrigated agriculture, the majority of the farms that require irrigation rely on natural springs, boreholes dug into the valley and small farm dams (Boshoff, 2005; Boshoff, 2008; Jansen, 2008; Kirkman, 2006). Ecotourism has increased in recent times (especially in and around the BMR) due to a decline in commercial agriculture as a result of

dropping market prices and increased minimum wages driving down profits as well as the attractiveness and remoteness of the area (Boshoff, 2005; DWAF, 2002; Knight, 2012). How people interact with the landscape and the environment is a core concept of any ESs approach as humans are the beneficiaries of such services (Gull, 2009). Different AESs will be evident where the land use varies, thus, it is imperative to understand area specific land uses, and in turn, what types of AESs are provided.

4.4. Summary

In summary, this chapter describes the reasons for choosing the Baviaanskloof catchment as the study area and briefly outlines some of the physical characteristics that are relevant in one way or another to the provision of AESs. The variety of land uses in the catchment are concisely described and their relevance to AESs is explained. Overall, this chapter provides a succinct and relevant description of the study area and relevant social and physical characteristics. Chapter 5 provides a detailed account of the research methods and study design used to achieve the aims and objectives of the study.

CHAPTER 5

RESEARCH METHODS AND STUDY DESIGN



Photo: J.M. Rawlins

5.1. Overview and Study Design Rationale

Figure 18 chronologically illustrates the four primary research methods used to address the aims of this study and effectively answer the research question. Together, these four methods were termed the ‘Ecosystem Service Value Chain Analysis (ESVCA) framework’ and were conducted via a two-phased approach. The first phase being ‘causal loop diagram development’ and the second phase being ‘causal loop diagram analysis’. The first phase of this approach included both the expert workshop and further professional and site verification, while the second phase was made up of the scenario and value chain analyses.



Figure 18: Study design flow chart.

In the fields of economics and natural science, case studies are often used to investigate the structure and function of specific processes, industries, regions and/or systems (Vihervaara *et al.*, 2010; Yin, 1994). McCutcheon and Meredith (1993) recommend the application of case study based research to narrow the gap between prescriptive advice and practical, workable answers for managers. Case studies are effective in developing new theoretical approaches and analysing unfamiliar circumstances (Voss *et al.*, 2002). The first phase builds on the CLD development methods for ESs described by Armah *et al.* (2010) and Pollard *et al.* (2014). The framework serves to test the applicability of using CLDs as a tool to conduct value chain analyses for AESs, using the Baviaanskloof catchment as a case study.

This two-phased approach was based on an adaption of the general three-step modelling process outlined by Costanza and Ruth (1998), the major system dynamics analysis processes discussed by Ford (1999) and Richardson and Pugh (1989) and the three-step value chain analysis processes put forward by MindTools (2014). Table 3 below illustrates how each step of the above mentioned processes are included in the two-phased approach of this study.

The first stage in the three-step modelling process involves creating a low resolution, high generality scoping model with numerous individuals interested in the research objectives (Costanza and Ruth, 1998). The second stage improves on the level of detail and complexity

of the research model through involving relevant experts that are concerned with the dynamics of the particular system under study and developing site-specific scenarios or policy options for the third stage of the process (Costanza and Ruth, 1998). The third stage is concentrated on generating scenarios and management options in the context of adaptive monitoring and feedbacks based on the analysis of the previous scoping and research models (Costanza and Ruth, 1998; Holling, 1978).

Table 3: Process steps that make up phase 1 and 2 of the study design.

		‘Major system dynamics analysis steps’ (Ford, 1999)	‘Three-step modelling process’ (Costanza and Ruth, 1998)	‘Value chain analysis steps’ (MindTools, 2014)
Phase 1	Expert Workshop	1. Problem definition and delimiting 2. Describing the underlying system 3. Model development	1. Low resolution, high generality scoping model	1. Activity Analysis
	Professional and Site Verification	4. Model Verification	2. Improve on level of detail & create site-specific scenarios	
Phase 2	Scenario & Value Chain Analyses	5. Modelling for analysis 6. Communication	3. Analyse scenario and management options	2. Value Analysis 3. Evaluation and Planning

The major system dynamics analysis processes include the following six steps: problem definition and delimiting, describing the underlying system, model development, model verification, modelling for analysis and communication (Ford, 1999; Richardson and Pugh,

1989). The three-step process to conduct a basic value chain analysis involves an activity analysis, a value analysis and, lastly, evaluation and planning (MindTools, 2014).

Due to the nature of the project, the complexity of the natural systems under study, knowledge gaps in available research and lack of sufficient quantitative data, this study uses an exclusively qualitative approach to inform the research goals and objectives. Patton (2005) emphasises the importance of naturalistic inquiry and inductive analyses that the qualitative research paradigm provides, generating narrative and systematic descriptions through the assessment of realistic case studies. Braun and Clarke (2013: 4) outline some broad differences between qualitative and quantitative research paradigms, highlighting the importance of qualitative modelling and subsequent analyses for complex systems.

The ESVCA framework can potentially be applied in different contexts and geographical locations, assuming the appropriate variables specific to the area are in/excluded in/from the CLD and the limitations and strengths of the approach are adhered to. Cresswell (2013) and Yin (2012) emphasise the importance of combining research design methods such as grounded theory and case study research. Adopting a combination of recognised research methods and processes improves the robustness of study design, the ease at which the design can be repeated and the relevance of the results to the field of study (Cresswell, 2013; Yin, 2012). The four general research methods outlined in Figure 18 and the various steps followed during each phase of the project shown in Table 3 are described and discussed below. This is followed by a discussion of the limitations and strengths of the study design and a general statement about the ethical considerations made during phase 1 of the project.

5.2. Phase 1: Causal Loop Diagram Development

5.2.1. Expert Workshop

An expert workshop was held where experts (see Appendix 1) with backgrounds in different disciplines ranging from environmental science to ecology and economics were engaged to achieve the following workshop objectives:

- a. Identify and describe final AESs that occur in the study area
- b. Identify and describe associated intermediate AESs

c. Develop CLD

The main aim of the workshop was to build dialog and facilitate a participatory approach towards developing CLDs with emphasis being placed on an agreed understanding of the interactions between the different AESs and associated intermediate services (Jafari *et al.*, 2008; Koca and Sverdrup, 2012; Wilson and Hoehn, 2006). McComb *et al.* (2006) describe the importance of participatory expert workshops in modern day multidisciplinary research. Workshops have been identified as highly effective tools for assessing complex systems such as aquatic ecosystems (Boshoff, 2005; Christie *et al.*, 2006). Fernald *et al.* (2012) further highlight the success of using workshops to facilitate conceptual modelling of complex environmental systems. Collaborative engagement of experts has been proven to successfully facilitate the determination of the feasibility of different environmental risk management scenarios (Ginsburg *et al.*, 2010; Nel *et al.*, 2011).

Deliberative processes make decision-making and research more transparent because experts can reach agreement through exploring transdisciplinary boundaries and allowing for the development of collective intelligence (Nel *et al.*, 2011; Parks and Gowdy, 2013). The USNRC (2005: 220) states “economists and ecologists should work together from the very beginning to ensure the output from any ecological model is in a form that can be used as input for an economic model”. The importance of utilising participatory expert workshops and professional verification approaches when developing qualitative economic models based on complex environmental-economic systems is clear.

Delimiting the scope of the workshop and the size and complexity of the CLD was essential as no concrete step-by-step methodology could be applied to this form of qualitative modelling process, nor has this framework been tested before within this context. The workshop was focused on facilitating the design and development of an adequately detailed CLD. The CLD was required to be sufficiently inclusive of natural and anthropogenic variables so that it would be suitably representative of the system. While at the same time, the CLD needed to be limited to only the most relevant and impactful variables affecting the final AESs so that it could be easily understood, analysed and communicated to a range of stakeholders and decision-makers. Ultimately, the scope and complexity of the CLD development process was decided through the consensus of the experts present at the workshop.

As illustrated in Table 3, the expert workshop went through a number of processes in order to achieve the desired objectives. The problem definition (step one of the major system dynamics analysis steps) was presented in the beginning of the workshop and then the participants were asked to discuss and delimit the problem so the aim and objectives of the study were aligned with the workshop objectives. The remainder of the steps and processes, outlined in Table 3, that were completed during the expert workshop are addressed and described under the individual workshop objectives below.

a) Identify and describe final aquatic ecosystem services

The first workshop objective was to identify, categorise and describe relevant, appropriate and sufficient final AESs that are aligned with the chosen definition of AESs and are agreed upon to exist in the Baviaanskloof catchment. Addressing the first workshop objective began with the participants being required to conduct ‘blind’ identification of up to five and no less than three final AESs. The ‘blind’ identification process was done individually without any group discussion or any prior information provided on the specific definitions of final and intermediate ESs. During this process, it became apparent that the expert attendees from different disciplines had widely divergent understandings of the concept of a final ES and how these services related to consistent evaluation of human benefits, which are common issues outlined by Boyd and Banzhaf (2007), Fisher *et al.* (2009) and Landers and Nahlik (2013). After this exercise, the specific definitions of final and intermediate AESs (below) were presented and discussed along with the method to distinguish between them and the CICES framework for categorising and classifying final ESs.

Final AESs:

Biophysical functions, processes or conditions of aquatic ecosystems that are utilised (actively or passively) to provide benefits that directly contribute towards the welfare of one or more human beneficiary.

Intermediate AESs:

Biophysical functions, processes or conditions of aquatic ecosystems that only provide benefits to humans indirectly through effects on other final ecosystem services first.

The rigorous, four-rule methodology for distinguishing between final and intermediate ESs described by Johnston and Russell (2011) was adopted during the first workshop objective to ensure a sound understanding of the differentiation between the two concepts. The four rules are defined by Johnston and Russell (2011: 2245-2247) as:

(1) Willingness to pay – “for biophysical outcome h to serve as an ecosystem service for beneficiary j , changes in h must influence the welfare of beneficiary j , so that a fully informed, rational beneficiary j would be willing to pay for increases in h rather than go without.”

(2) Natural outputs – “for biophysical outcome h to serve as an ecosystem service for beneficiary j , h must represent the output of an ecological system prior to any combination with human labour, capital or technology.”

(3) Direct benefits – “For endpoint h to serve as a final ecosystem service for rational beneficiary j , the beneficiary must be willing to pay for increases in h , assuming that all other ecosystem outputs and conditions $i \neq h$ are held constant.”

(4) Services to all beneficiaries – “an ecosystem outcome h can also simultaneously represent both a final service to beneficiary j and an intermediate service to another beneficiary $n \neq j$.”

The CICES framework was briefly presented to illustrate the different classifications and options for final ESs and how they relate to other ESs. However, it was made clear that this framework is not comprehensive and area specific final ESs may differ significantly. Once the nuanced differences between the two definitions were made clear with respect to the aim of the study and objectives of the workshop, and the four rules for distinguishing between final and intermediate ESs (Johnston and Russell, 2011) were understood, the participants were asked to methodically begin with a cooperative identification of final AESs that occur in the Baviaanskloof catchment.

The workshop was based upon a demand side approach towards the development of the CLD as it began with the final AESs (the good or service that is demanded by a particular beneficiary) and worked backwards through the various intermediate services towards the processes and functions that contribute towards the supply of the final AESs. Each participant was required to systematically identify a particular beneficiary group (e.g. farmers), then a specific benefit that contributes to the welfare of the beneficiary group (e.g. water for

irrigation) and, lastly, the final AES that provides this benefit (e.g. water provision). This identification procedure was done a minimum of three times for each participant. The selection of the final AESs for the model was based on the relevance of the services for the study area, their level of integration into formal markets, whether the quality and/or quantity of delivery can be altered by potential management scenarios and related information/knowledge availability (Landuyt *et al.*, 2014). This ES identification process is aligned with the general frameworks for ES identification described by COWI (2014) and Landers and Nahlik (2013).

It was emphasised that each final AES identified should ideally have obvious and clearly definable causal relationships with their associated intermediate services/processes and be easily distinguishable from other ESs. The aim of this is to simplify further the next two workshop objectives. Many of the definitional issues surrounding ESs noted by Haines-Young and Potschin (2014) are addressed through the robust approach described by Johnston and Russell (2011) as well as the causes of ambiguity in classifying and quantifying values associated with intermediate versus final ESs.

Once at least three final AESs were identified and agreed upon to occur in the study area then one of them was chosen as a 'flagship' service would be used for an extended analysis. The purpose of this was to account for the complexity of the system and the diagram, which was to be developed in such a way that a general value chain analysis could be conducted on all three final AESs. Additionally, this would allow a significantly more detailed analysis to be carried out on the 'flagship' service, as a more detailed illustration of the associated intermediate processes would be incorporated into the CLD. This was purely for the purpose of limiting the size of the diagram as CLDs have the potential to become extremely cluttered and complex, which render them ineffective management and decision-making tools. The 'flagship' service was identified based on the potential size of the value chain, the ease of conducting a sufficiently comprehensive analysis and the degree of integration of the service into formal markets. A structured methodology of this nature was chosen in an attempt to remain consistent and allow for comparisons between approaches towards ES identification and classification. Despite not including any form of ecosystem valuation, clear distinctions still need to be made between final and intermediate ESs as these findings could potentially be used in further research.

b) Identify and describe associated intermediate aquatic ecosystem services

The second workshop objective was to identify, categorise and describe the associated intermediate processes, variables, conditions or AESs that directly or indirectly affect the previously identified final AESs. To address this workshop objective the concept of the ES Cascade (Figure 2) was presented alongside the diagram illustrating the relationships of aquatic ecosystem processes to ESs (Figure 7) (Brauman *et al.*, 2007; Haines-Young and Potschin, 2011). Together with the aforementioned definition of intermediate AESs, this entrenched the understanding of the relationships between final and intermediate AESs and thus, facilitated the identification of the underlying functions and processes (intermediate services) that contribute to the generation of the final AESs.

Adhering to the methodology developed by Johnston and Russell (2011), a minimum of two intermediate AESs were identified for each of the three final AESs. These were considered first tier intermediate services as they directly affected the final AESs. Once a consensus was established over these intermediate services then the further intermediate services were identified only for the 'flagship' service. This process began with the identification and description of as many second tier intermediate services (those that directly impact the first tier intermediate services) as possible. Eventually third and fourth tier intermediate services were identified for the 'flagship' final AES. Due to interlinking nature of the CLD and the complex environmental-economic system under study, more than four tiers of intermediate ESs are evident if analysed in a traditionally linear fashion (explained in section 5.3.2).

All of the intermediate services/processes/conditions/variables were distinguished into either environmental or anthropogenic variables. An environmental variable is any process or variable that occurs without or is independent of any direct human influence (e.g. rainfall) while an anthropogenic variable is a process or variable that occurs as a direct result of or is dependent on human agency (e.g. water abstraction).

During this phase of the workshop two specific steps outlined in Table 3 needed to be addressed, namely 'describing the underlying system' (Ford, 1999) and 'activity analysis' (MindTools, 2014). Identifying and describing all of the associated intermediate ESs is akin to describing the underlying processes of the system. While an activity analysis involves brainstorming all the activities and processes that contribute towards the final outcome in

the value chain, in this case this refers to the final AESs. Hence, all of the supporting services make up the contributing activities and processes.

c) Develop causal loop diagram

The third and final workshop objective was to develop the CLD. Vensim PLE version 6.3 software was used to graphically create the diagram and display all of the linkages in real time during the workshop. All of the final and intermediate AESs that were previously identified were first created in Vensim and added to the diagram. The next step was to systematically identify and define the complex causal relationships and linkages between all of the final and intermediate AESs, which had now been added as variables into the diagram. These relationships needed to be defined on the diagram in a binary manner (either positive or negative). However, it was also necessary to create a narrative around the nature of each of the relationships to gain a better understanding of their differences in spatial and temporal scale. This insight was imperative for identifying and developing the different scenarios (discussed in section 5.3.1.)

This procedure was conducted in a step-by-step manner where the relationships between the first tier intermediate AESs and the final AESs were defined first. Then the relationships between the different tiers of intermediate AESs were defined one by one while continuously making note of any factors that may affect the quantity, quality, timing and location of the affected service (Brauman *et al.*, 2007). Once most of the relationships were defined and added to the diagram, the floor was opened to suggestions as to what other variables should be included in the diagram. It was made clear that only the most relevant variables should be included to ensure the diagram is complex enough to accurately represent the natural system while also simple enough to be used as an effective management tool. An open dialogue remained until the participants were satisfied with the structure, function, relationships and variables in the diagram.

Ultimately, through this process a relatively low-resolution, high generality scoping model was designed that is specific to the study area. Hence the 'model development' step (Ford, 1999) and the 'low resolution, high generality scoping model' were completed as outlined in Table 3. The model graphically illustrates the relationships between all the different types of variables within the diagram regardless of whether it is a final AES or a fourth tier

intermediate AES, even final AESs affect one another directly. Vensim is a very simple and powerful software tool capable of extending an analysis of this nature when supplied with relevant and sufficient quantitative data that defines the relationships between the different variables. Pollard *et al.* (2013) provide a critique of Vensim and a comparison to similar software, however, Vensim was more than suitable for the purpose of this study.

5.2.2. Professional and Site Verification

Formal meetings/interviews were set up with several professionals and relevant informants from around the country (see Appendix 1) to verify, improve and supplement the CLD where needed. As with the expert workshop, the participants were from a variety of academic and professional backgrounds with first-hand experience in the Baviaanskloof catchment. However, these meetings were much less structured and an open dialogue was propagated around the realism and accuracy of the diagram. This facilitated a significant amount of knowledge input into the diagram in terms of specifically defining each variable and potential units of measurement (Appendix 2).

During the formal meetings, the CLD was tested and the general flow logic of the different variables and linkages were examined. Disturbances, shocks or changes to the system were suggested and analysed to verify the robustness of the diagram and make it as suitable for the purpose of the study as possible. When referring to the various CLDs, the term 'analyse' is used exclusively in a qualitative sense and involves a broad assessment of specific drivers of change on the system and any associated deductions that can be made from the system changes. Grösser (2012) and Kirkwood (2013) emphasise the importance of presenting such qualitative models of the system to interested and affected parties because secondary information about the system itself is often key to improving the effectiveness and accuracy of the diagram/model. Interacting with relevant and informed stakeholders can often produce valuable information not available in the literature and goes beyond the issues focused on by specialists (COWI, 2014).

A field visit to the study area was also conducted which allowed for a first-hand experience of the aquatic system and the different services that it provides. Being able to engage with academics, professionals and local people who live in the catchment allowed for the conception of the four different scenarios, which would be analysed using the model. The

study area visit was extremely important as the scenarios needed to be as realistic and specific as possible. Many of the outcomes of these scenarios are generally understood and have been experienced by people in the catchment over many years; hence, this provided the opportunity to determine whether the CLD could model a similar outcome with the input of a particular scenario. In most cases, the last step of any qualitative model development or ES identification process involves on-site examination and local expert judgement (COWI, 2014). However, Saladin *et al.* (2012) highlight how highly case- and site- specific ESs are and thus there is always significant potential for a lack of information and data to exist in the study area.

The fourth step outlined by Ford (1999) in Table 3, i.e. 'model verification', was completed during this stage of the research as far as possible. The model could not be quantitatively justified but was verified by relevant professionals and local expert judgement specifically focusing on the identified scenarios presented below in section 5.3.1. This process improved the level of detail of the model and facilitated the creation of the four different system scenarios to be assessed in more detail. Thus, effectively completing step two outlined in Table 3 by Costanza and Ruth (1998).

Ultimately, this research component contributed immensely to the reliability, accuracy, realism and robustness of the CLD when applied to the specific scenarios. Moreover, it facilitated the development of each scenario to ensure they were area specific, realistic and have the potential to benefit people living in the catchment when modelled and managed in the future. Developing a model that easily and straightforwardly presents such a large amount of information in a graphical way allowed for the scenario and value chain analyses to go forward in phase two.

5.3. Phase 2: Causal Loop Diagram Analysis

5.3.1. Scenario Analyses

Four different scenarios in the Baviaanskloof were analysed using the CLD, namely fire, drought, land-use change and catchment restoration. Fire and drought are naturally occurring phenomena while the remaining two scenarios are anthropogenically induced. All four scenarios occur over very different time scales. In general, fire has an immediate impact,

drought is a long-term phenomenon, land-use change is short to medium-term and the catchment restoration scenario occurs over a medium to long-term scale. These four scenarios were chosen as they are realistic situations within the Baviaanskloof catchment and thus, will have representative and practical management and decision-making outcomes. Furthermore, the varying temporal scales over which the scenarios occur provide a broad array of circumstances to test the CLD as a tool to analyse AES value chains.

Scenario analyses are a common approach for analysing trade-offs between ES delivery and their implications for human well-being, most notably in the Millennium Ecosystem Assessment (MA, 2005; Pirard *et al.*, 2010). Scenario analyses are often an integral component of various ES frameworks for management and decision-making (Guswa *et al.*, 2014; Walker *et al.*, 2002). Carpenter *et al.* (2006) and Carpenter *et al.* (2009) highlight a number of challenges that ESs create for scenario analyses surrounding ecological feedbacks, uncertainty and regime shifts. The scenario analysis approach adopted during this research project is novel in the sense that it directly incorporates ES CLDs as the primary focus of the analysis.

Using CLDs to analyse the various scenarios facilitates the identification of relevant ripple effects within the system. These ripple or multiplier effects are evident in how a single change affects multiple system components, directly or indirectly. However, a traditional approach towards a ripple and/or multiplier analysis does not apply to the context of this study, due to the public nature of the ESs and the associated absence of formal market data (input cost structure and market clearing prices). Hence, a broader interpretation of 'ripple effects' was required to complete the analysis. The CLD developed during phase 1 of the project illustrates the positive and negative forward linkages of the ESs that it models, which allows for the assessment of the ripple effects that result from external influences or "shocks" to the system. These external "shocks" were simulated through the case specific scenarios.

Before the scenario analyses could be carried out, an accurate and detailed description of each scenario was needed. Each scenario could have a multitude of impacts on the system, depending on the individual circumstances and associated ripple effects. For example, a fire has the potential to directly affect natural vegetation, crops and soil stability. These four

analyses attempt to simulate potential opportunities and challenges that could affect the provision of and markets for the final AESs.

The scenario analyses were conducted using a supply side approach, each assessment began with the disturbance to the system (e.g. fire) then logically followed the impact of the disturbance through the various linkages until the nature of the impact on the final AES could be determined. The outcome of these systematic processes are CLDs that illustrate the effect of the disturbance on the system through highlighted arrows, only the direct and indirect impacts of the particular scenario are included. The immediate, short and long-term impacts are shown by different colours of arrows in an attempt to compensate for the diagrams limited ability to illustrate differences in temporal scales. Specifically, 'short-term' refers to impacts that occur within days or weeks of the disturbance while 'long-term' considers months to years in duration. It is important to note that it is not possible for one variable to have a shorter term impact on another variable than the original impact on itself. This is coherent with the purpose of the scenario assessments, to illustrate the impact of selected scenarios on the system as a whole.

The final element of the scenario analyses involved describing the outcome of the scenarios on the provision of the final AESs in positive or negative terms. Ultimately, the purpose of assessing the effects of different "shocks" to the system was to provide the necessary tools and information to be able to conduct an alternative value chain analysis for each of the final AESs.

5.3.2. Value Chain Analyses

Value chain analysis has become an increasingly valuable method to develop a more detailed outlook of the various stages and variables involved in taking a good or service from raw materials, to production and through to the consumer (Schmitz, 2005). The ultimate aim of this process is to construct a better understanding of how and where people and organisations are positioned within particular value chains and to identify potential leverage points and opportunities for improvement (Baum, 2013; IMA, 1996; Kaplinsky and Morris, 2000). The concept of value chain analysis was developed to identify potential areas of competitive advantage for private sector entities through maximising the activities in any of the relevant variables in the value chain (IMA, 1996; Kaplinsky and Morris, 2000; Porter, 1985;

Schmitz, 2005). This objective is analogous to improving the output or increasing the provision of final ESs. However, public goods and the markets for these goods and services are not constrained to meeting the financial bottom line; this is not the only objective. Other objectives other than creating a competitive advantage can be achieved, such as improving the ecological and environmental quality of the natural system that provides these services to improve the standard of living of the ES beneficiaries.

On the other hand, value chain analyses can be used to illustrate the underlying processes and functions that contribute to the provision of the ESs (or raw materials) that private enterprises are so reliant upon. If it were clearly understood how investment into specific components of public good value chains could improve the sustainability and/or provision of a service that an industry is reliant upon to function and remain competitive, then this would act as an incentive for private and public investment into the sustainable management of these natural systems. The result of this process is to develop extended, more inclusive value chains that formally integrate ESs into established markets for goods and services that depend on the provision of these services.

A holistic approach is required to assess complex natural systems that incorporate multiple variables and nonlinearities, however, traditional value chain analyses are atomistic and linear in nature. This, in theory, makes it impossible to perform a traditional, linear type of analysis, as there is no defined starting point for an ES based value chain. Before the alternative value chain analyses of the identified final AESs could begin, an attempt was made to create a traditional, linear value chain for water provision. This utilised the intermediate AESs identified in the CLD created during the first phase of the project. Each intermediate ES was traced backwards from the final AES, this process was completed for four tiers of intermediate AESs. This process practically demonstrated the theoretical issues surrounding the application of a traditional value chain analysis to a suite of ESs that form part of a complex socio-ecological system. Emergent properties of these systems ensure that shocks to the system affect variables at different spatial and temporal scales and at different magnitudes depending on the circumstances of the impact and ripple effects. This process emphasised the need for scenario analyses to provide a starting point for a value chain analysis.

Similar to the method of the expert workshop, the alternative value chain analyses were conducted using a demand side approach (working from the final AES backwards) to first identify individual linear causal pathways of cause and effect from the scenario analysis diagrams. These causal pathways or individual value chains flow from the change in the system (the starting point identified through the scenario analyses) through to the impact on the final AES. When there are multiple causal pathways affecting the same variable in opposite directions then no robust conclusions could be drawn as to the impact on the final AES. However, informed deductions could be made about the time scale of the impact and potentially the magnitude of the different impacts if the catchment dynamics are well understood. Through this process, different examples of value chains were identified that are the most relevant to the objective of a management and/or decision-making process; these are coherent with the concept of 'value factors' described MindTools (2014).

The identification of individual value chain examples allowed for linear visualisation and evaluation of how changes to the system affect the provision of final AESs. From these examples, potential management options were explored for each of the scenarios to provide future planning opportunities to improve the positive impacts or mitigate the negative impacts on the provision of the final services. Accordingly, this allowed for a description of the general limitations and strengths of the approach. Thus, effectively concluding the 'value analysis' and 'evaluation and planning' steps outlined by MindTools (2014) in Table 3. It is evident that the 'modelling for analysis' and 'analysing scenario and management options' steps termed by Ford (1999) and Costanza and Ruth (1998) respectively were addressed throughout the scenario analysis and value chain analysis processes. Lastly, the 'communication' stage described by Ford (1999) is achieved by writing and publishing the information regarding the methodology and research results and making it available to relevant stakeholders.

5.4. Limitations and Strengths

The novel nature of this research has resulted in numerous practical and theoretical limitations and challenges for the ESVCA framework approach. On the other hand, the combination of different methods and approaches implemented in this study has provided a

theoretically and practically sound study design, which has the potential to be easily repeated in the future.

The composite study design was developed in an attempt to address the various limitations associated with the three individual methods as far as possible (Table 3). However, combining these approaches is constrained from a theoretical standpoint in terms of the scale and resolution of application. For example, it is difficult to determine a precise level of detail when applying the 'describing the underlying system' and 'low resolution, high generality scoping model' steps during the professional workshops. Thus, the parameters for each of the steps outlined in Table 3 were defined during the workshop stage to ensure uniformity through the multiple approaches. Further theoretical limitations surrounding the application of a value chain analysis via a system dynamics model of AESs have been outlined in chapters 2 and 3.

The expert workshop was limited in terms of the number of people involved, only experienced professionals in related disciplines that have undertaken some form of research in the study area before were considered. However, being situated in a university environment relatively nearby to the study area provided the best opportunity to find the most appropriate participants. Delimiting the goals and scope of the workshop itself was limited by the amount of time the participants were willing to donate for the purpose of the research. The inclusion of the professional and site verification component attempted to address the limitations associated with the expert workshop. However, the field trip was also limited in terms of time and the availability of relevant people in and around the study area.

The terrain of the Baviaanskloof catchment is very rugged and the dirt roads require a 4x4 vehicle and a significant amount of time to traverse. This hindered the extent of the catchment that could be explored in the limited time available. The Vensim PLE software is provided free to researchers, which made the physical creation of the CLD simple. While the qualitative nature of the study design allowed for the system to be modelled in a relatively short period.

The chosen scenarios are very different from one another in terms of their impact on the system over time. This facilitated a comprehensive investigation of the CLD's strength as a tool for value chain analysis. The method requires the identification of linear casual pathways from a CLD illustrating a disturbance to a system, this limits the application of the value chain

analyses because they can only be conducted in response to a particular disturbance/scenario. At the same time, this could be viewed as an advantage as it focuses the analysis on a particular outcome and considers specific influences in isolation. Although the study design is limited in a number of ways, the method can be easily adjusted to incorporate new research techniques involving ESs.

5.5. Ethical Considerations

Through the duration of this research project, it was ensured that all requirements of the Rhodes University Research Ethical Standards Policy were observed and adhered to. The research methods were presented to the Department of Economics and Economic History's ethical standards sub-committee and were given ethical clearance. All workshop participants and informants were provided with ethical consent forms that stated the necessary details regarding the aim and objectives of the project and manner in which the information and data would be used and distributed. The form provided the opportunity to remain anonymous or to have only specific information about oneself published with the results. Additionally, it was stated that all participants/informants right to exit participation from the study at any point was reserved. Feedback in the form of a summary report of the outcomes of the project was offered to each participant/informant. The objective and impersonal nature of the information required from the participants and informants did not require the research project to be approved by the Rhodes University Ethics Committee.

5.6. Summary

In summary, this chapter provides detailed descriptions of the two phases of the study design that make up the ESVCA framework, their purpose, context and the theory behind them. The reasoning behind the choice and development of the different methods is explained. Specific limitations and strengths of the study design are discussed in practical terms and the measures taken to address the limitations are put forward. Based on the theory presented and referred to in this chapter, chapter 6 demonstrates the outcomes of the methodological approach and explains how this addresses the primary aim of the project.

CHAPTER 6

SCENARIO AND VALUE CHAIN ANALYSES



Photo: J.M. Rawlins

6.1 Rationale

The outcome of the ‘causal loop diagram development’ phase of this study is illustrated in Figure 19; three earlier versions are illustrated in Appendix 3. The figure is a CLD that demonstrates the various processes and functions that contribute towards the provision of three final AESs, namely water provision, flood attenuation and aquatic ecosystem health. In particular, this CLD distinguishes between final and intermediate AESs, environmental variables and anthropogenic variables. A descriptive summary presented in Appendix 2 provides specific descriptions and categorisations for each of the final and intermediate AESs demonstrated in Figure 19. The name of each variable was kept as succinct as possible to avoid clutter in the diagram. This outcome effectively addresses the first specific research objective, i.e. development of an AES based CLD.

The CLD demonstrates how the complex array of intermediate AESs affects the provision of these final services in a multi-dimensional snapshot. Each causal linkage (arrow) qualitatively indicates the relationship between the two variables that it connects. These linkages effectively demonstrate the forward linkages in the value chain components of the final AESs, thus, facilitating the application of scenario and value chain analyses in the ‘causal loop diagram analysis’ phase of the project.

The final AESs included in the CLD were chosen based on their level of integration into the formal market place, current understanding of the service and whether they fit into the different CICES categories. In order to address the aim of the study, the three chosen final AESs varied significantly according to these decision criteria. Water provision was identified as the ‘flagship’ service because the market for water is well established and understood, there is also a specific CICES classification for this service. Flood attenuation is a process that is well understood and there is a specific CICES classification for it, however it is much less integrated into formal markets than water provision. Lastly, aquatic ecosystem health is understood in very general terms, but is not directly referred to under any of the CICES classifications and is the least integrated into formal markets out of the three.

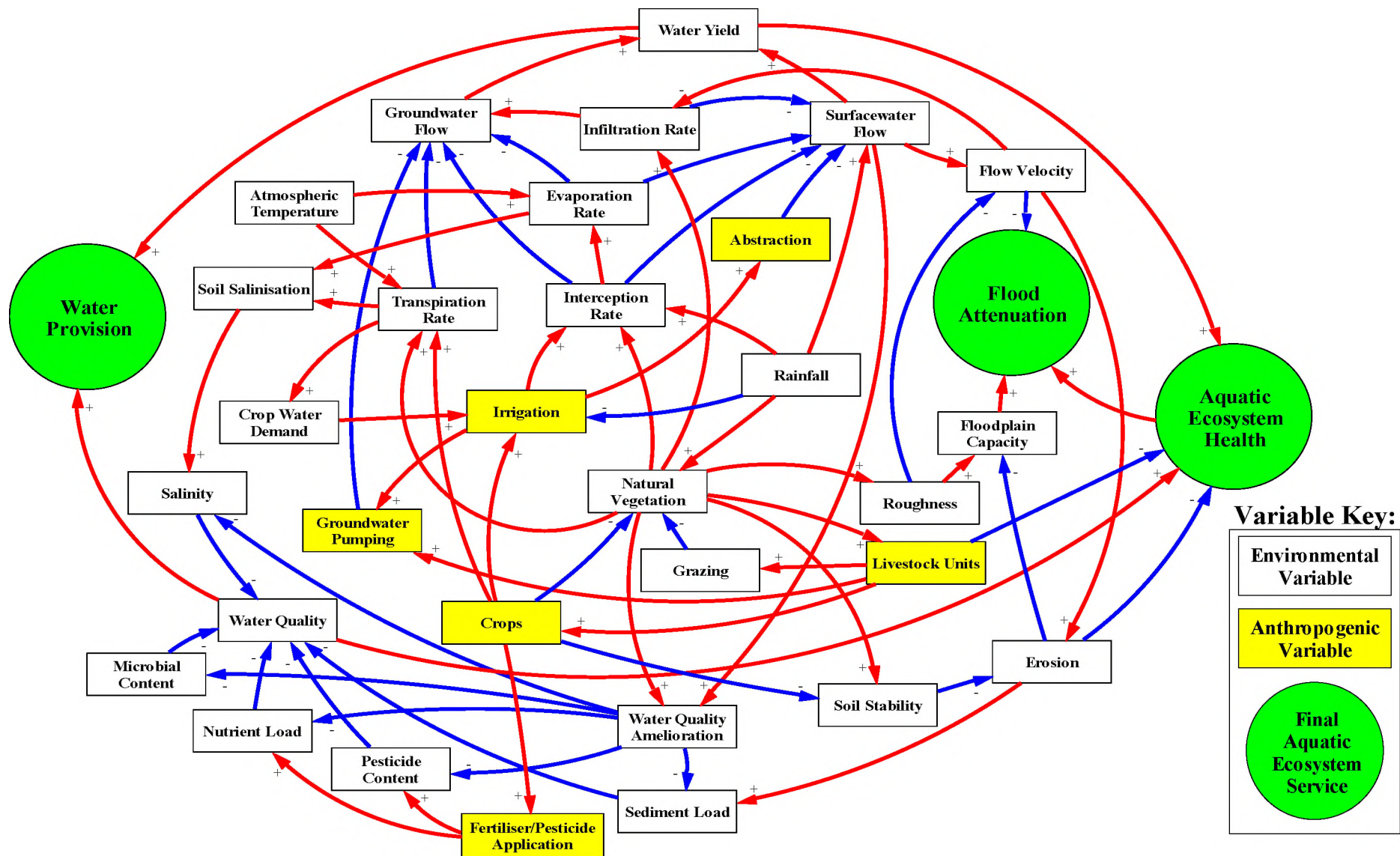


Figure 19: Causal Loop Diagram Illustrating Select Aquatic Ecosystem Services, Processes and Functions in the Baviaanskloof Catchment.

Due to the complex and stochastic nature of the system under study and the tiered structure of the intermediate ESs, no positive or negative feedback loops were identified. Kirkwood (2013) and Sterman (2000) identify this type of approach as open loop thinking or 'pejorative thinking'. Moreover, graphically depicting these feedback loops would not contribute towards conducting a successful value chain analysis on the final AESs as any feedback scenarios would be captured in the scenario analyses.

The suitability of CLDs to assess the value chains of AESs in South Africa is determined through the scenario analyses. Each scenario allows for the identification of linear causal pathways (individual value chains) that clearly demonstrate how various disturbances influence the provision of the final ES at hand. Thus, allowing for the identification of potential challenges and/or opportunities in these individual value chains, regardless of how integrated the service is into formal, existing markets. Simultaneously the ripple effects of the various disturbances to the system are illustrated, specifically focusing on their impact on the provision of the final AESs. However, although the ripple effects in the system are analysed during these analyses it is not possible to analyse the knock-on effects over long time scales that result from the changes induced by a particular scenario. This is because the scenario analyses isolate one or two variables that change as a result of a specific system change and then examine the ripple effects of the disturbance on the system. For example, the ripple effects of fire will be investigated in terms of a decrease in natural vegetation in the catchment; an indirect knock-on impact of this could be the regrowth of the vegetation over time. This type of knock-on effect is not considered, as the purpose of the analysis is to determine the impact of a change on a system in isolation. Quantitative system dynamics modelling will be required to enable modelling of such knock-on effects.

The scenario analyses were designed to pave the way forward for the alternative value chain analyses to be conducted for each individual final AES. The different scenarios simulate how positive or negative impacts on the system influence the system as a whole and how this affects the provision of the final AESs in a multi-dimensional way. Then the value chain analyses can focus on the final AESs and identify individual value chains from each scenario to be investigated in a more traditionally linear manner. It must be noted that the CLD illustrated in Figure 19 was developed specifically for the purpose of this study and is not intended to be comprehensive.

The different approaches that make up the ESVCA framework and their outcomes are presented and discussed below, focusing on determining the suitability of CLDs as an alternative tool to assess the value chains of AESs. This chapter presents detailed descriptions of the four scenario analyses and examples of potential value chain analyses for each of the final AESs identified in Figure 19. Lastly, general strengths and limitations of the ESVCA framework is presented and discussed.

6.2. Scenario Analyses

6.2.1. Fire

The fire scenario is an example of a short-term event that simulates a once off fire event in the Baviaanskloof catchment. Fire is a very common disturbance in the Baviaanskloof and plays an important role for veld management in the area (Boshoff, 2005; Glenday, pers. comm. 2015). The area is governed by a natural fire regime, which is an essential part of many ecological cycles and assists with the propagation of many endemic *Fynbos* species that occur in the area (Booyesen and Tainton, 1984; Glenday, pers. comm. 2015; Kruger, pers. comm. 2015). This scenario, in particular, replicated a fire that was severe enough to significantly reduce the amount of natural vegetation in the area without affecting crops or livestock. While simultaneously not being hot enough to have any type of effect on the soil properties that support the vegetation. Hence, the only variable that was directly affected through the fire scenario in the CLD was natural vegetation.

Analysing this event as a scenario illustrates the various ripple effects that transpire from this event over three aforementioned time scales (immediate, short-term and medium/long-term) as exemplified in Figure 20a. The figure demonstrates how the disturbance variable, fire in this case, will have an impact on all three of the final AESs. It is clear that the fire would decrease the ability of the system to attenuate floods immediately afterwards. The general aquatic ecosystem health would decrease in the medium to long term because of the fire. The fire scenario seems to have a generally negative impact on water provision, as it would decrease the water quality in the short to medium term. However, the impact on water yield

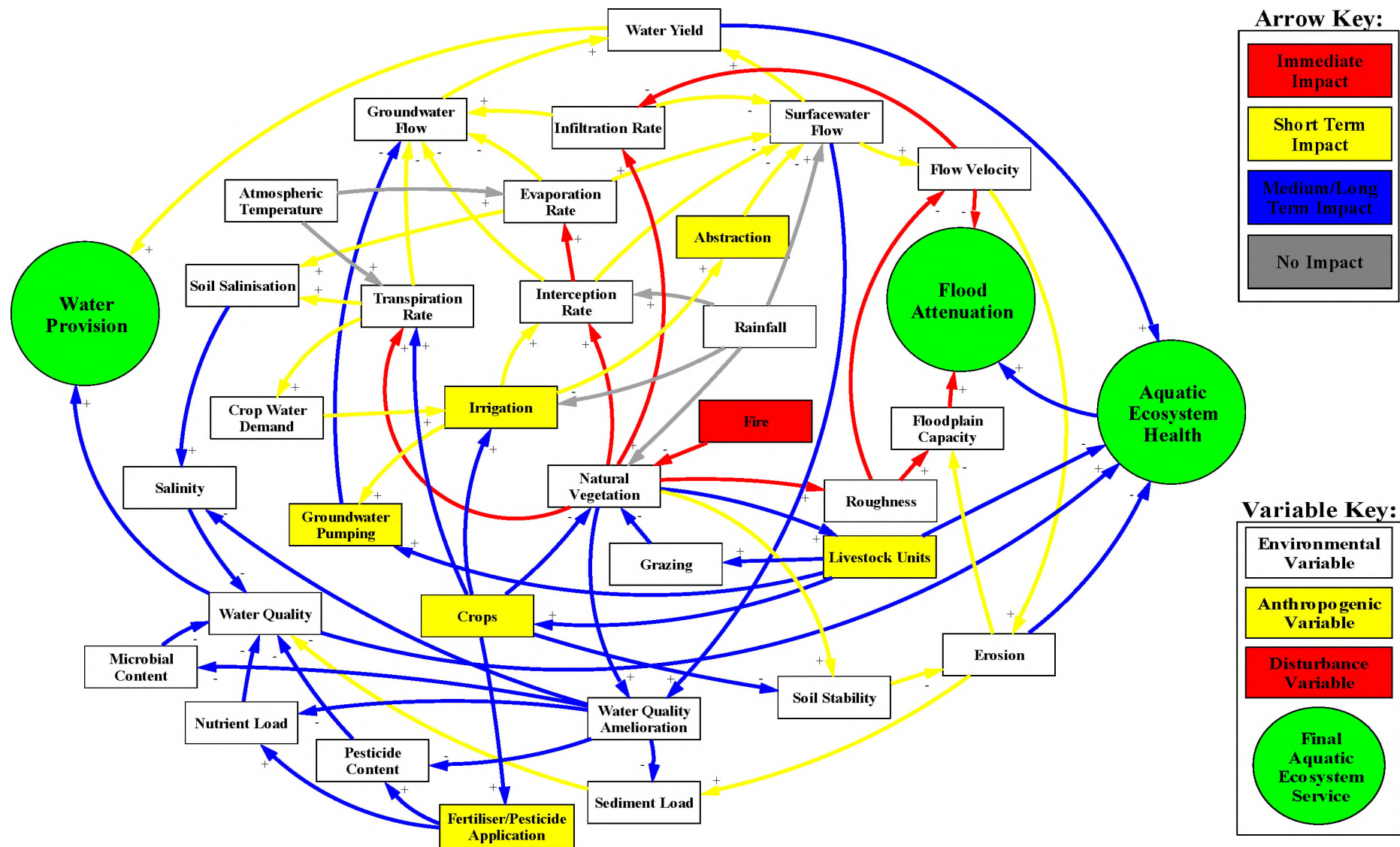


Figure 20a: Fire Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment.

cannot be determined as groundwater flow increases and surface water flow decreases as an indirect result of the fire. As previously mentioned this outcome does not take into consideration any knock-on effects such as the regrowth of vegetation over time, which could counter these negative impacts.

6.2.2. Drought

The drought scenario is an illustration of a long-term occurrence that attempts to simulate a decrease in rainfall over a long period. Drought is a relatively common occurrence in the Baviaanskloof, there is often less than a 100 mm of rainfall in a year (Kruger, pers. comm. 2015; Mander *et al.*, 2010; Mendelsohn, pers. comm. 2015). Weather patterns around the world are being influenced more and more by climate change, a largely natural phenomenon accelerated through human agency. As a result, severe weather events such as droughts and floods are becoming more prevalent and intense (Houghton, 2004; IPCC, 2013). Although the majority of the farmers in the Baviaanskloof have access to groundwater for irrigation and livestock, they are still reliant on rainfall during certain times of the year (Kruger, pers. comm. 2015).

Figure 20b illustrates the impact of drought on the system highlighting the different time scales of impacts. Despite drought occurring over a very long period, the impacts were considered to occur over a range of time scales as if the drought occurred at one point in time. This is purely for the purpose of the analysis and illustrates the varying time scales of the impacts associated with a decrease in rainfall. From Figure 20b, it is evident that a drought would have a negative impact on water provision within an immediate to short time frame. This is logical with regard to water yield; however, less surface water reduces the dilution capacity of the system and thus decreases its ability to ameliorate water quality. Hence, the quality of water in the system would be decreased in the short to medium term. The general aquatic ecosystem health is also decreased over time but due to the broad nature of this concept, the effects would only be evident in the long run. Interestingly, a drought would initially have a positive impact on the system's ability attenuate floods as indicated by the immediate decrease in flow velocity. However, over time the decrease in natural vegetation will decrease the roughness and thus the floodplain capacity of the system and ultimately the system's capacity to attenuate floods. It must be noted that no absolute deductions can be

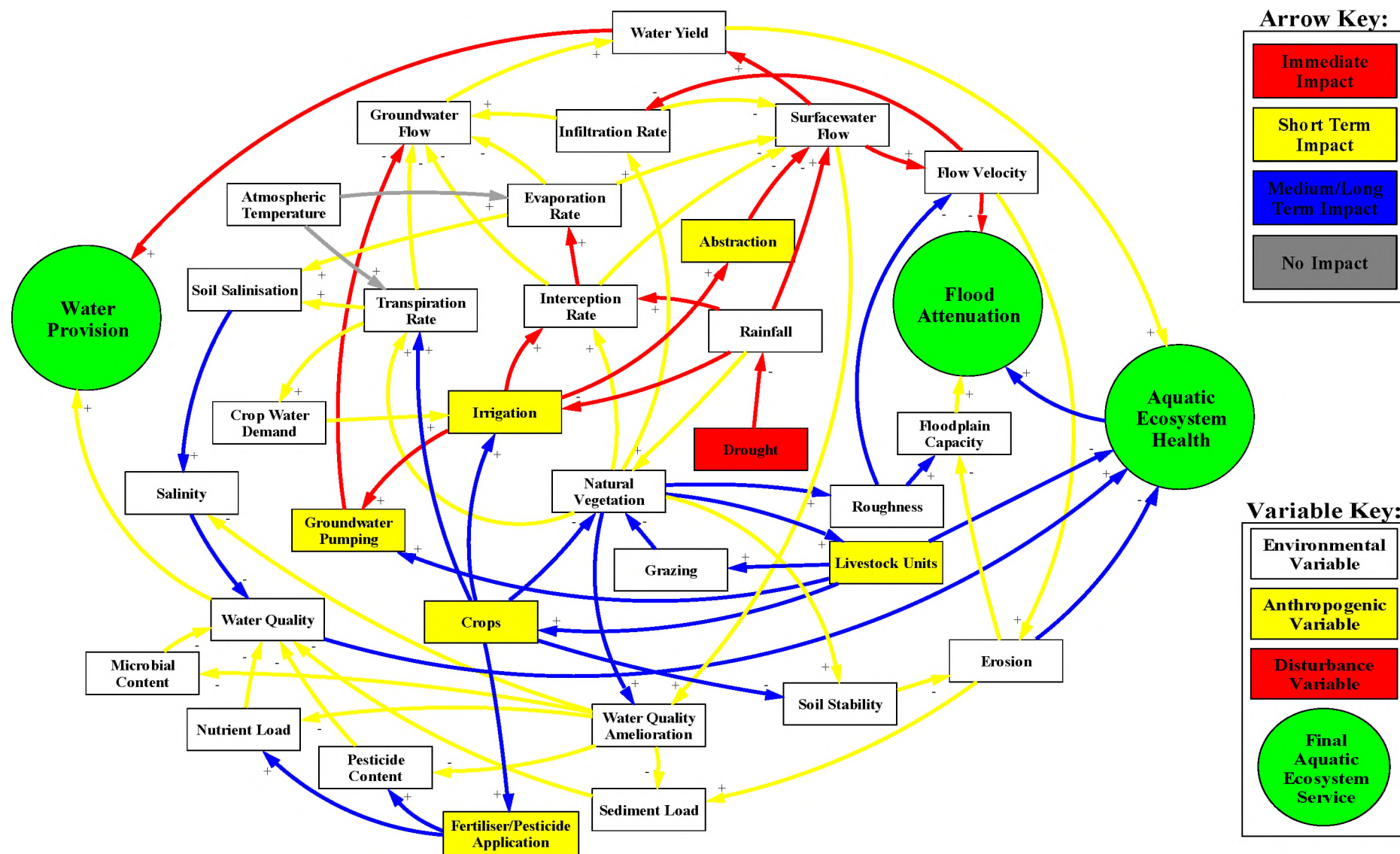


Figure 20b: Drought Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment.

made regarding the direction in which flood attenuation would be affected due to the qualitative nature of the model; this will be discussed further during the examples of basic value chain assessments (section 6.3).

6.2.3. Land-use Change

The land-use change scenario is an illustration of a proposed business plan by two organisations known as the Four Returns Development Company and Livinglands (FourReturns, 2014). The aim of the project is to assist local farmers in converting from growing fodder crops for their livestock to growing lavender to harvest the essential oils. As lavender has a relatively low water demand, this will reduce water use in the valley. Accordingly, this will also reduce livestock numbers, as farmers will be focusing on harvesting lavender rather than rearing livestock. All the seed and plant material will be supplied by the development corporation at no cost to the local farmers. The ultimate aim of the project is to slow the flow of water through the catchment system (promote diffuse flow) and create an alternative income stream for vulnerable farming communities.

The project will maintain the same area of land under cultivation and thus the only two variables that will be affected in the CLD are crop water demand and livestock units. It is clear from Figure 20c that a decrease in the crop water demand and number of livestock units in the Baviaanskloof catchment because of the change in land-use will have a positive impact on all of the final AESs. The amount of water available at an adequate quality within the system will logically increase as less water is being used for irrigation etc. The ability of the system to attenuate floods will probably decrease in the short to medium term because of increased flow velocity but once again will improve in the long run as the roughness and floodplain capacity will increase. Less livestock and more water available will also increase the general aquatic ecosystem health in the medium to long run. These outcomes highlight how the aims and objectives of this project are theoretically supported through this causal model.

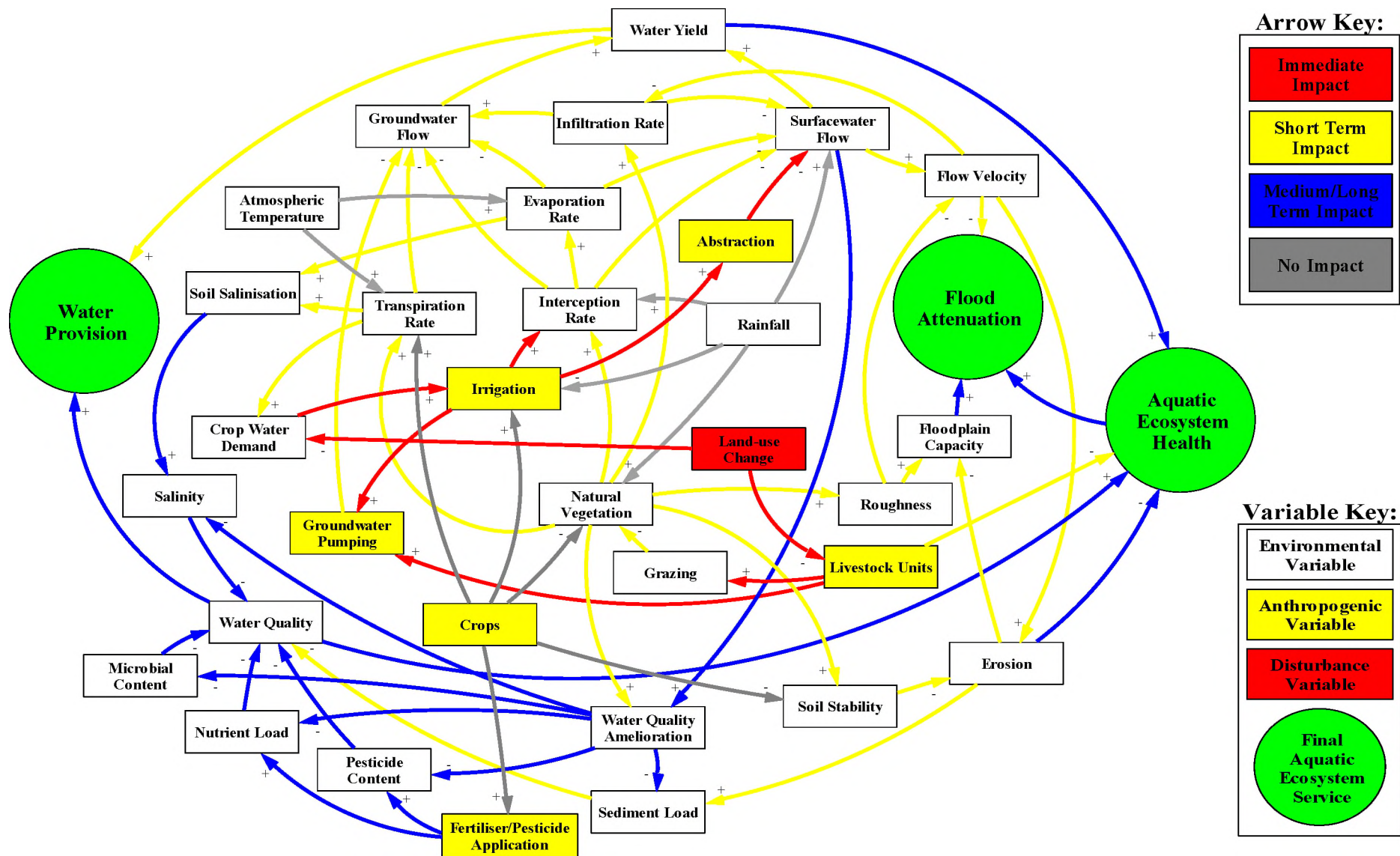


Figure 20c: Land-use Change Scenario Analysis Causal Loop Diagram for Aquatic Ecosystem Services in the Baviaanskloof Catchment.

6.2.4. Catchment Restoration

The catchment restoration scenario is a demonstration of some of the restoration activities that are currently being performed in the Baviaanskloof catchment. The use of restoration as a management tool to improve AES health and provision has grown rapidly over the last decade (Palmer *et al.*, 2014). The STRP and WfW are the primary proponents of these restoration initiatives (Boshoff, 2005; Powell; 2009). These activities include the building of weirs, gabions and small dams as well as the restoration of alluvial fans and the planting of *Portulacaria afra* ('spekboom') in degraded areas (Illgner and Haigh, 2003; Kruger, pers. comm. 2015; Powell, 2009). The ultimate purpose of these activities is to promote diffuse flow of water throughout the catchment in order to retain as much water as possible for as long as possible.

Considering all of these restoration activities broadly, the scenario simulates the impact of increasing the surface roughness within the catchment and the associated impacts throughout the system. Figure 20d illustrates how catchment restoration activities can have a mixed impact on the provision of the three final AESs as an indirect result of increasing catchment roughness. With regard to water provision, the decrease in flow velocity will increase infiltration and thus increase groundwater flow while decreasing surface water flow, resulting in an ambiguous effect on water yield. Similarly, an immediate decrease in sediment load is met by a longer-term decrease in dilution capacity, which have opposite effects on water quality. Hence, one cannot deduce the nature of the impact on water provision over the long-term. The system's ability to attenuate floods will be explicitly increased in the immediate term. However, the impacts on aquatic ecosystem health are once again ambiguous as it should improve in the short to medium term due to decreased erosion but increase in the long term due to decreased water quality.

6.3. Value Chain Analyses

The scenario analyses illustrated above outline some of the complex interactions within the system that arise from naturally and anthropogenically induced changes and represent them in a holistic and inclusive manner. The outcomes of these analyses are complex in themselves and not very easy to use for management and decision-making purposes.

Figure 21 illustrates an attempt to create a traditional style, linear value chain for water provision from the processes and functions highlighted in the CLD in Figure 19. The purpose of this exercise is to illustrate how it is theoretically possible to create comprehensive, linear, traditional value chains that model complex systems. However, through ‘forcing’ the traditional approach towards value chain analysis for an intricate ecosystem it is clear that it is not a practical solution. The value chain only traces four tiers of causal linkages and already it has become too cluttered to be efficient and/or user-friendly. Considering a complete value chain of this nature would trace back every tier of causal influence that could have some form of impact on the final ES, the chain would become exceptionally complicated and convoluted. Especially considering how the natural feedback mechanisms (illustrated via the blue interlinking arrows) would make the linear chain loop back on itself and further complicate the model. If only the linear relationships were considered and the interlinking causal relationships were ignored, then the chain would not be an accurate representation of the system. Once again emphasising how traditional, linear value chain analysis approaches would not be an appropriate management tool capable of effectively representing stochastic systems. Hence, the use of CLDs and scenario analyses to allow for an alternative approach towards value chain analyses to be conducted for these systems.

A structured, alternative approach is required to conduct a viable value chain analysis for ESs. Firstly, the investigation process would involve determining the objective of a particular system intervention and or mitigation initiative (e.g. to reduce the impact of fire on a system’s ability to attenuate floods). Then from the scenario analysis CLDs, the relevant linear causal pathways (value chains), flowing from the fire disturbance to flood attenuation, would need to be identified. Assessing these individual value chains will identify the ideal areas to intervene to best reduce the negative impact of fire on flood attenuation. Potential leverage points in these value chains can be single or multiple environmental and/or anthropogenic

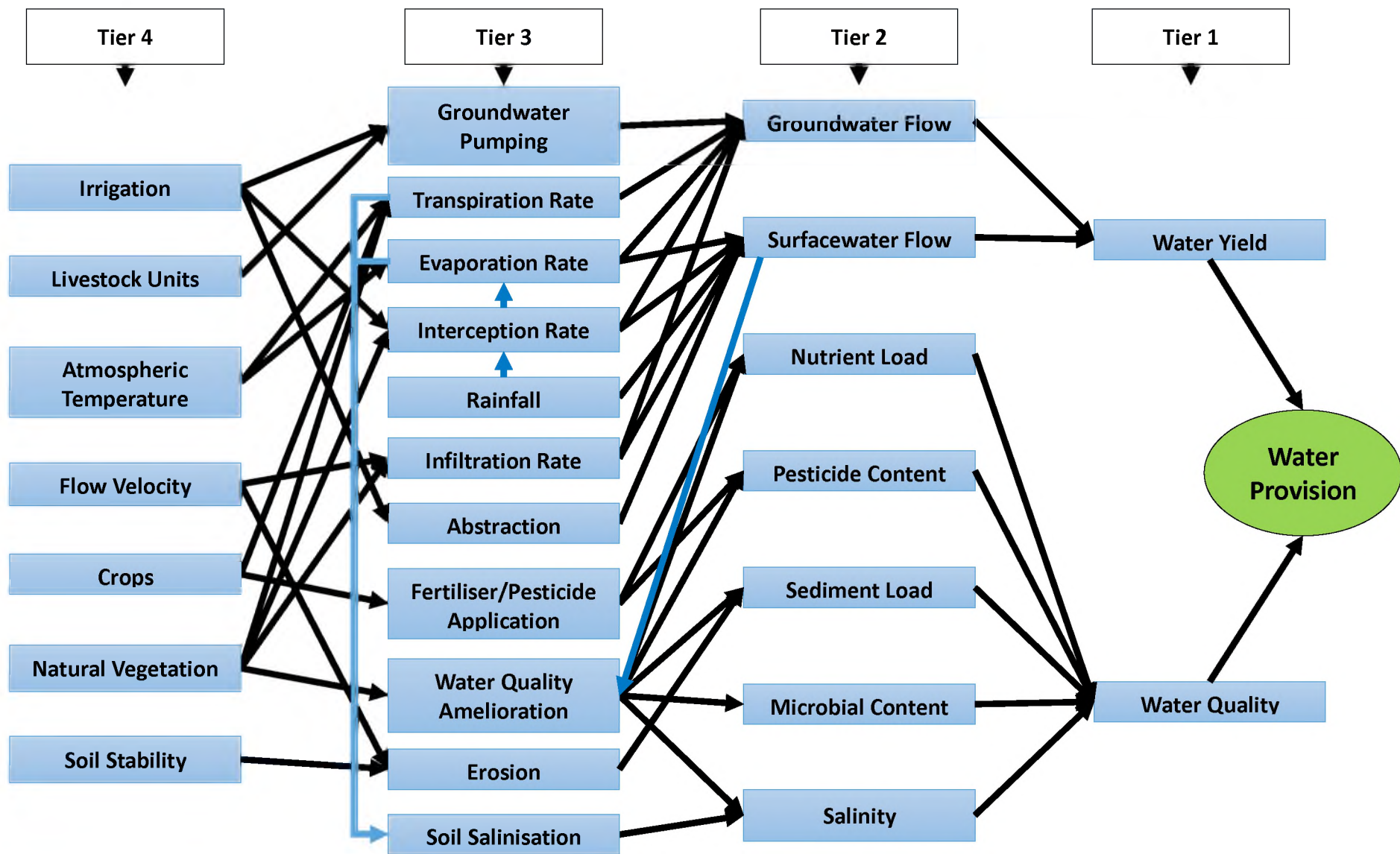


Figure 21: Four Tier Traditional Value Chain Example for Water Provision.

variables and/or any of the linkages between them. The example of fire impact mitigation for flood attenuation will be discussed in section 6.3.2. This process ultimately assists decision-makers and managers to make the most out of opportunities and/or mitigate/prevent potential threats before it is too late. Alternative value chain assessment examples are illustrated through individual linear causal pathways and discussed for the three final AESs identified in the original CLD below. The blue arrows in figures 22a, 22b and 22c illustrate a decrease while the red arrows show an increase in a particular variable as a result of the change in the previous. Black arrows indicate that the outcome is indeterminate.

6.3.1. Water Provision

Figure 22a presents four individual linear causal pathways that illustrate an example of how each of the scenarios affects water provision. As previously mentioned, these individual linear causal pathways are used to demonstrate specific and relevant sections of the value chains of the final AESs. The value chain is broken down into these individual pathways to create manageable units, which can then be analysed. The first linear pathway showing the impact of fire is indeterminate due to the contrasting impact of a decrease in infiltration rate on groundwater and surface water. Both ground and surface water variables are included in this value chain to illustrate the contrasting impact and how this could be a potential problem when selecting individual causal pathways. Thus, it is critical to use the scenario analyses with the CLDs first to identify potential contrasting impacts on the same variable and then narrow it down to linear causal pathways of interest to the objective at hand. This allows logical, coherent and consistent (not absolute) inferences to be made around the occurrence of these contrasting impacts. For example, it is fairly obvious to deduce that there will be no increase or decrease in the water yield as a result of the decrease in the infiltration rate but rather a change in the timing and position of the flow (surface water tends to flow faster than groundwater). However, contrasting impacts on the same variable are often an issue of magnitude (i.e. which impact is greater). In these cases, it is not possible to determine whether the increase or decrease is greater as the relationships/impacts are not quantified.

To provide an example of how these value chains can be analysed to assist management and decision-making, an objective needs to be identified first. For example, the Gamtoos Irrigation Board or Water User Association is trying to devise a mitigation plan for the impact of an

impending drought on the provision of water in the Baviaanskloof catchment, with specific focus on the water yield. Figure 22a provides two basic examples of linear causal pathways that show how drought decreases the amount of surface water and thus water yield and provision. Only considering these pathways, irrigation and abstraction are the most obvious leverage points for attempting to mitigate the impact of drought on water yield. Rainfall and surface water flow can technically be altered through human influence but will be much more difficult. For example, alternative water storage methods could be adopted to reduce abstraction or alternative irrigation infrastructure and methods could be investigated to reduce the amount of water loss. On the other hand, adopting the land-use change scenario could be considered as an option as it decreases the crop water demand and thus the amount of irrigation required as well (Figure 22a). Private firms interested in maintaining high water yields or mitigating against the impacts of a decreased water yield (e.g. companies that sell bottled water) could also use this approach to identify potential leverage points for improvement or possible mitigation initiatives, these would have financial benefits for the firm in the long run.

This type of analysis is not limited to analysing a single linear casual pathway, analysing multiple pathways simultaneously will often provide more potential options. However, the choice of linear pathways to analyse is completely dependent on the requirements and aims of the desired intervention as well as other constraints such as budget, etc. The same analysis could potentially be done directly by using the CLD and without breaking down the CLD into individual linear causal pathways. However, the impact flows become very difficult to follow when considering the entire system at the same time. Developing a structured, step-by-step process is easier to repeat and being able to provide a simplified decision framework to managers and decision-makers from the relevant professionals is the most efficient approach.

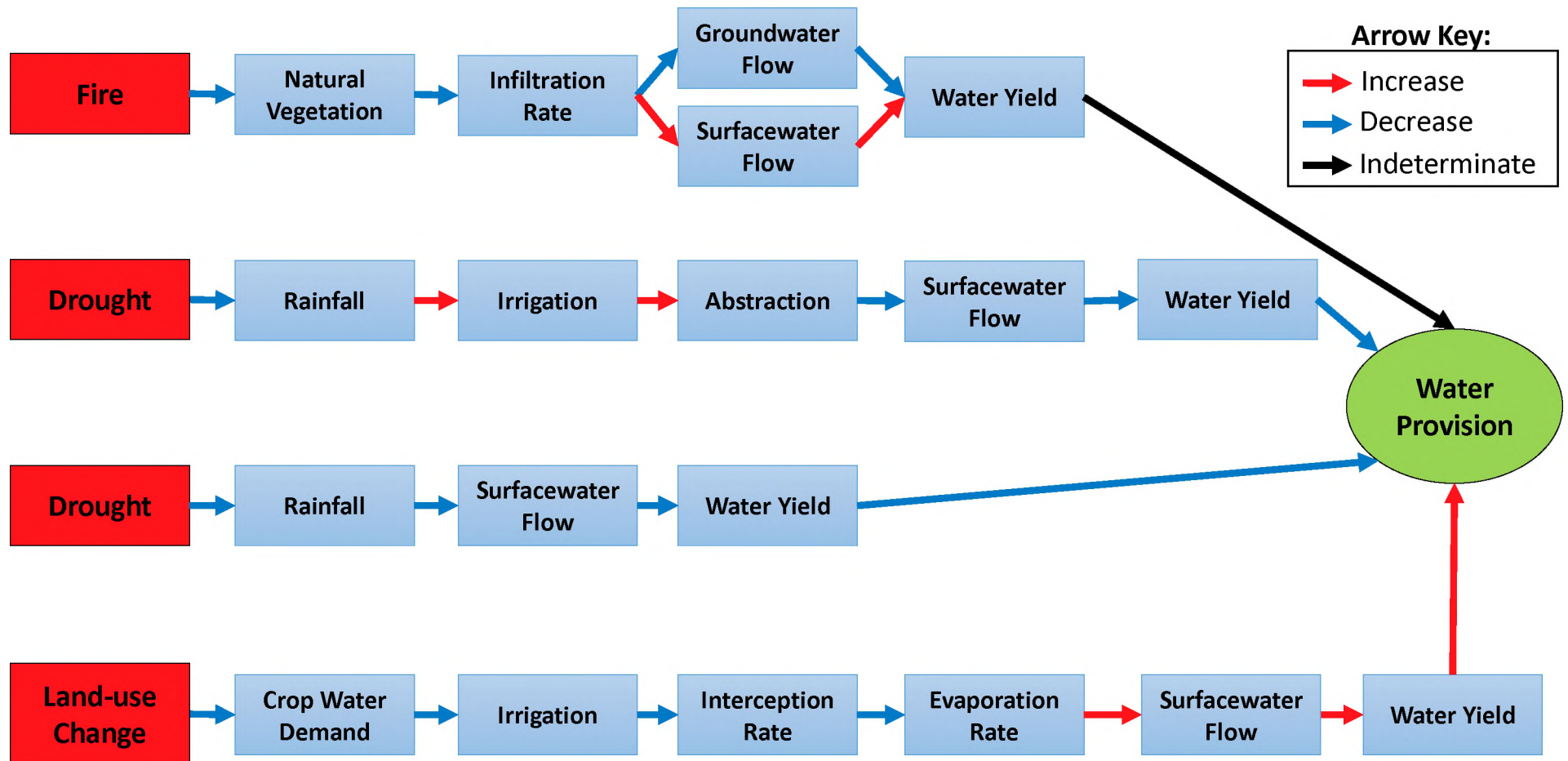


Figure 22a: Examples of Linear Causal Pathways Affecting Water Provision.

6.3.2. Flood Attenuation

Figure 22b illustrates three examples of linear causal pathways that represent the effect of fire and catchment restoration on the system's ability to attenuate floods. For example, a private insurance company is interested in conducting a value chain analysis for this particular final AES to identify possible intervention measures that will reduce the impact of fire on the ability of the system to attenuate floods. Fire provides a potential financial threat to an institution of this nature by reducing the ability of the system to attenuate flooding, as a result more flood damage claims are put forward and the company is required to pay out a larger amount of money. Thus, if a cost-effective option to reduce the effect that fire has on the ability of the system to attenuate flooding can be identified, this will increase the profitability of the company in the long run. Flooding is a relatively common occurrence in the Baviaanskloof, making this a realistic outcome for investment companies (Jansen, 2008; Van Der Burg, 2008).

Considering the first two linear value chains presented in Figure 22b, the loss of natural vegetation because of a fire event indirectly decreases the floodplain capacity and increases flow velocity; both of which have a negative impact on flood attenuation. Thus, the apparent leverage variables for intervention would be natural vegetation, roughness and infiltration rate. Propagating and promoting the growth of fire resistant indigenous plants would theoretically reduce the loss of vegetation resulting from fire and increase system roughness and the infiltration rate (Booyesen and Tainton, 1984). This could be supplemented with a geological survey that identifies the most efficient and effective areas to promote infiltration (e.g. closest to the phreatic or saturated zone). Alternatively, the aforementioned catchment restoration scenario, directly and over a relatively fast time scale, improves the flood attenuation capacity in the Baviaanskloof catchment (Figure 22b). If these specific relationships between fire and flood attenuation highlighted in Figure 22b could be quantified, then it would be possible for the firm to conduct a benefit-cost analysis to determine whether the investment would be financially viable or not. Aside from other apparent benefits such as improved water retention in the system, which will benefit local residents etc. This method could also be adopted to develop disaster management plans by government with the aim of preventing the loss of life and infrastructural damage.

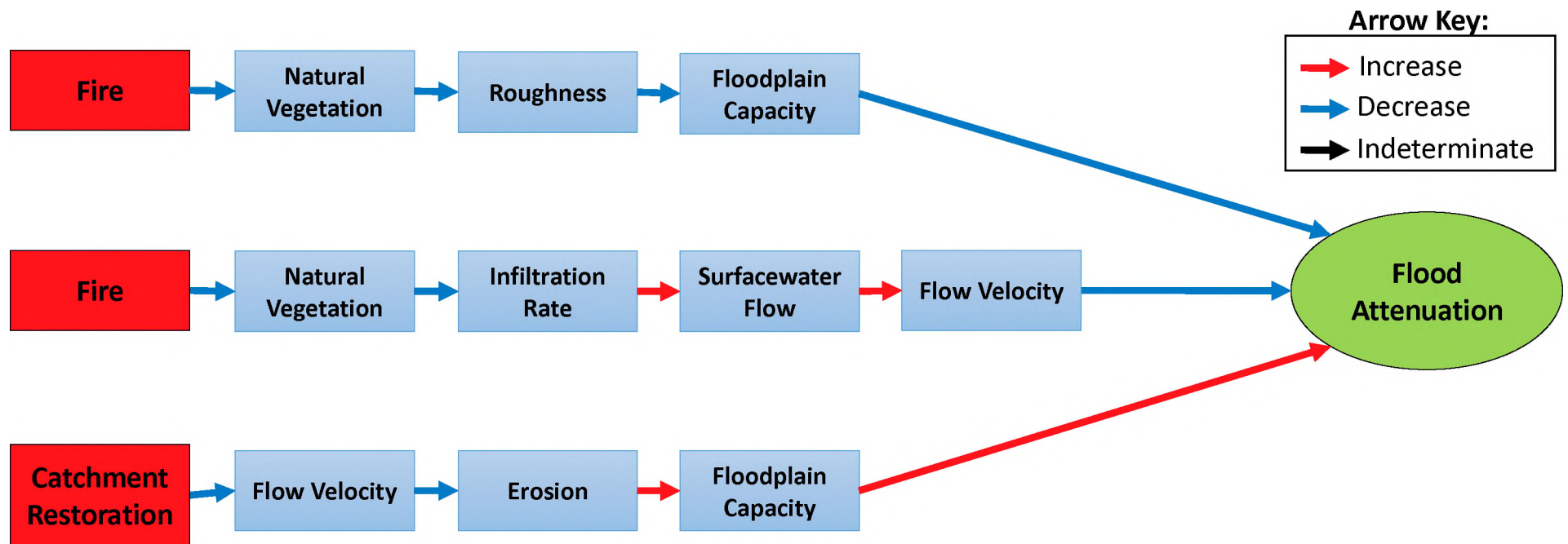


Figure 22b: Examples of Linear Causal Pathways Affecting Flood Attenuation.

6.3.3. Aquatic Ecosystem Health

Aquatic ecosystem health is the least integrated into formal markets out of the final AESs being assessed. Hence, it is problematic to develop a scenario that would promote private investment into the restoration and/or conservation of the catchment. It would also be the most difficult of the final AESs to quantify. Figure 22c presents linear examples of how each of the four different scenarios could have an impact on the general aquatic ecosystem health of the Baviaanskloof catchment. An example of where aquatic ecosystem health could be the target of a sustainable management intervention or public/private investment is in the conservation domain. Conservationists and recreationalists in the area are direct beneficiaries of a healthy ecosystem and thus, would be interested in developing sustainable management initiatives that would contribute towards maintaining ecosystem integrity.

When analysing Figure 22c, it is evident that an increase in erosion and a decrease in water yield are the two first tier impacts that cause a decrease in aquatic ecosystem health. It is clear from the catchment restoration and land-use change linear pathways that these scenarios would counter the negative impact of drought and fire on aquatic ecosystem health. Other potential mitigation scenarios exist that focus specifically on erosion, catchment roughness and groundwater pumping. As previously mentioned, planting fire resistant plants would increase the roughness of the catchment and thus, decrease flow velocity and then erosion. From a private investment standpoint, the ecosystem that is being improved would need to generate some form of income to justify any conservation efforts. For example, if there were a private tourism concession within the catchment, then a healthy aquatic ecosystem would have numerous positive impacts for tourists and conservationists alike. Consequently, a concession of this nature would be incentivised to restore degraded areas of the catchment and promote land-use change upstream to receive the benefits illustrated in Figure 22c. A multitude of potential mitigation scenarios would improve the provision of aquatic ecosystem health while simultaneously having numerous other positive impacts on the system as a whole. It is important to understand how general aquatic ecosystem health directly and indirectly affects the other final AESs (Figure 19) that have more easily identifiable marketable attributes. This type of final service is often considered as an indirect benefit of

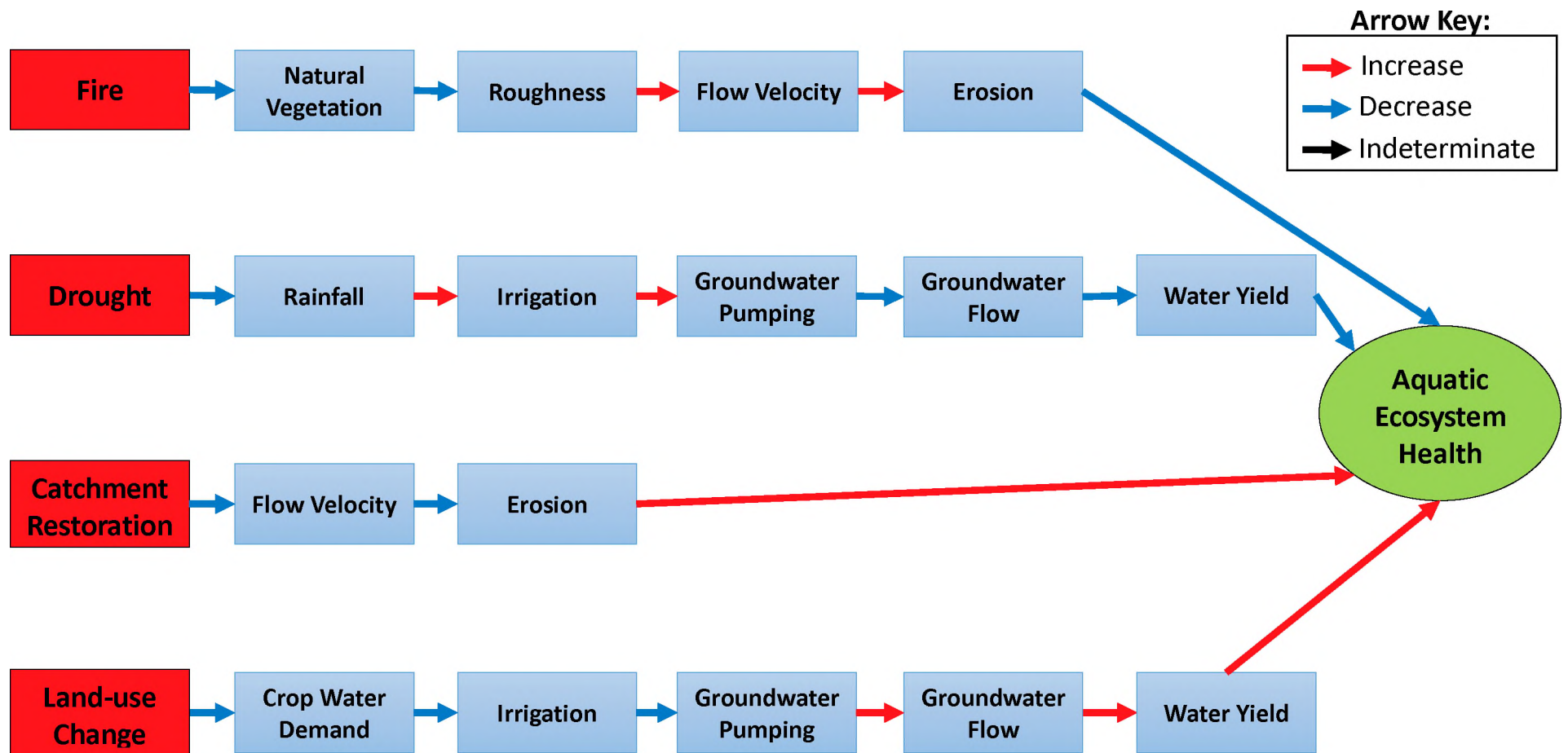


Figure 22c: Examples of Linear Causal Pathways Affecting Aquatic Ecosystem Health.

management and investment endeavours, but nonetheless has the potential to have significant positive impacts on the system as a whole.

6.4. Strengths and Limitations

There are a multitude of strengths and limitations associated with the approaches adopted by the ESVCA framework presented and discussed above. Thus, it is especially important to acknowledge and understand these to ensure this alternative approach towards value chain analysis is conducted in appropriate circumstances and is not over or underestimated in terms of its capability. It is imperative to use CLDs and their associated linear causal pathways together, to ensure a holistic analysis that considers all potentially conflicting impacts.

The decision-making tools generated from the ESVCA framework promote systemic thinking within management endeavours, a need that is widely published across many disciplines (Sterman, 2000). The framework attempts to close the gap between traditional, linear economic thought and the complex systems they attempt to model and/or value. The tool is more inclusive than current environmental management models, as it aims to model environmental as well as anthropogenic components. The model that the framework uses is predictive in nature, allowing pro-active strategies to be implemented through the identification of potential future system threats via relevant scenario analyses.

The predictive appeal of the model and the ability to graphically demonstrate the causal linkages and individual value chains that contribute towards the generation of marketed and non-marketed goods and services makes the ESVCA approach attractive to public and private sector investors. As previously highlighted in sections 6.3.1 – 6.3.3, the assessment framework has the potential to incentivise private investment into the sustainable management of ecosystems by presenting the various system processes that contribute to the generation of marketed goods and services. This will not only assist with the integration of ESs into formal markets but will result in numerous additional benefits associated with healthy functioning ecosystems such as improved aquatic species diversity and abundance. The market making mechanisms for ESs will be assisted through this type of analysis as it outlines linkages between final and intermediate ESs.

One major advantage that arose from the ESVCA framework is that it facilitates an in depth analysis of not only how specific disturbances (e.g. fire) will affect the provision of a desired final AES (e.g. flood attenuation) but also the potential intervention and/or mitigation measures (e.g. catchment restoration). In turn, this facilitates the analysis of trade-offs between various ESs as their individual inputs and potential to add value are compared and scrutinised (Brauman *et al.*, 2007). The method evidently has the ability to address the issues of changing ecological feedbacks, however, currently cannot account for potential regime shifts (Carpenter *et al.*, 2009).

In theory, the ESVCA approach can be effortlessly adapted to a variety of geographical regions in a manner that is comparable to that of a 'benefits transfer', thus making it an extremely versatile method for ES assessment and analysis (Barton, 2002; Plummer, 2009). The framework delivers a set of dynamic, synergistic tools that are parametrised towards beneficiaries and end-users. Ultimately, providing a suite of powerful tools that facilitate more informed decision-making with a plethora of options available for further development.

Although this approach is incredibly useful, there are also many limitations which restrict its application in different circumstances. The primary limitation associated with a tool of this nature is its inability to accurately account for changes in spatial scales because it has to be calibrated to a specific area. Despite being addressed to a certain degree by the use of different colour arrows in the scenario analysis CLDs, the approach cannot fully account for changes in temporal scale either. Due to the complex, stochastic and dynamic nature of natural systems it is incredibly difficult to delimit the optimum size of a CLD that incorporates sufficient complexity to accurately represent the system while making it simple enough to use as a management tool. For addressing the aim of this project, the optimal size of the CLD was gained through a consensus with relevant professionals, for other purposes the general size and level of inclusivity may vary significantly.

Without empirically defining the causal relationships between the different variables, it is not possible to determine the magnitude of impacts. Hence, when there are two opposite impacts on one variable one cannot categorically deduce the direction of the causality. Similarly, each relationship between different variables in the system had to be classified exclusively as either positive or negative. Thus, the ESVCA framework would not be able to capture the impacts of

differing magnitudes of change if they result in a change in the causality of the relationship. For example, a small increase in one variable could have a positive impact on the next variable, but a large increase in that same variable could result in a negative impact on the next variable. This was addressed as far as possible through the expert input and area-specific definitions (Appendix 2) which are assumed to occur within a particular threshold.

The non-linear and ever changing nature of aquatic systems made it virtually impossible to distinguish between and identify all of the relevant stock and flow variables at a uniform scale. Every one of the variables considered in the system change over time, thus, measurements would need to be taken over a specific time interval to provide meaningful results in respect to flow variables. To assess or measure these variables at a specific point in time (snapshot) would significantly reduce the accuracy of subsequent analyses. As a result, no variables were classified as either stock or flow variables and no causal linkages were classified as conserved or information flows in this study, despite the majority of variables defined in terms of their occurrence over time (Appendix 2). This was the main reason for not developing the CLDs from SFDs initially. Nevertheless, this did not have any impact on the logic flow of the diagram or its ability to realistically represent the system and potential system changes. Although, it must be noted that it is a deviation from the traditional approach towards developing CLDs outlined in Chapter 3.

The scenario analyses were limited to analysing one scenario at a time. This is a result of numerous conflicting impacts on the same variable occurring when too many influences are included at the same time. As the relationships between the different variables are not quantified, one cannot deduce the outcome of conflicting impacts on one variable. Considering this, along with the CLD's limited ability to represent changes in time, associated knock-on impacts cannot be simultaneously analysed. For example, knowledge of an impending drought might change human behaviour in terms of water use, which may affect the outcome of drought on the system etc.

The CLD is in itself limited in terms of the number of variables because of the challenges with representing spatial scale. Technically, variables can continue to be added to eventually include microscopic processes that in some way or another impact the final ES, but this is clearly not practical. Limiting the CLD to only the most relevant and important processes to

the final AESs attempts to capture the major impacts throughout the system. These are also the most important variables to manage with limited funds and time. Although this approach is limited from many angles, further research and development of this method could address many of these issues.

6.5 Summary

In summary, this chapter illustrates the findings from the first phase of the project and the subsequent analyses of these findings that make up the second phase. Using the CLD, four different scenarios specific to the Baviaanskloof were outlined and analysed to demonstrate the effect of a 'shock' or disturbance on the system. Then the alternative approach towards value chain analysis was presented in a logical step-by-step manner that drew on the scenario analyses to develop examples of individual linear casual pathways. Lastly, the strengths and limitations of the approach in general were discussed to provide a more realistic understanding of where and when this framework should be adopted and what needs to be done next. Based on this, Chapter 7 begins by providing and discussing some recommendations for future research and practical applications of the ESVCA framework/approach. Current knowledge gaps around the area of research are analysed to elucidate the way forward. Then an overview of the entire study is presented, which contextualises the findings within current literature. Lastly, conclusions are made concerning the outcomes, aims and objectives of the study.

CHAPTER 7

RECOMMENDATIONS AND CONCLUSIONS



Photo: J.M. Rawlins

7.1. Recommendations

7.1.1. Future Research

It is evident from the broad array of objectives, concepts and methods adopted during this study that there are numerous potential future research endeavours with the potential to generate original and relevant content to improve on the ESVCA framework developed throughout this study. Before the framework can be formally integrated into catchment scale management and decision-making, further research will need to be undertaken in an attempt to corroborate the findings of this study and address some of the fundamental limitations as far as possible. Ostrom (2007) describes how the relative success and/or failure of ES maintenance and management endeavours is highly context-specific, no policy or intervention is a panacea. Potential future research pathways that would address these requirements include further case studies that incorporate a completely different suite of intermediate and final ESs that have different levels of alteration and market integration. This would be particularly effective in urban environments as they feature highly altered ESs that would illustrate a significant contrast from the 'largely natural' Baviaanskloof catchment (Bolund and Hunhammar, 1999; Cilliers *et al.*, 2012; Palmer *et al.*, 2014). An inquiry of this nature will possibly elucidate the geographical and physical versatility of the tool, contribute towards further development of the framework and corroborate the rigour of the method. Appendices 1-3 are included to illustrate a more detailed description of the CLD development phase that could be used to assist future research and applications of this approach.

Carpenter *et al.* (2009: 1311) states that "in any particular situation, available options must be evaluated, selected, implemented, tested, and then replaced or modified in an ongoing search for better outcomes". The aim of this study was to determine whether CLDs could potentially be utilised as suitable and effective tools for conducting value chain analyses for AESs. The qualitative nature of the research provides the framework and foundation for future quantitative research into application of the same tool. A fully inclusive system dynamics model, based on the ESVCA framework, which integrates quantitative modelling techniques to define the relationships between different variables is a realistic outcome of further research. This could include quantitative hydrological modelling, water quality modelling, human behaviour modelling as well as GIS scenario modelling to mention a few.

Quantified relationships between the different variables will enable the user to determine the magnitude of change in provision of particular AESs. Integrating behaviour over time graphs will address the issues of scale by incorporating dynamic response thresholds, delays, etc. Not only will this improve the ability of the model to accurately account for changes in spatial and temporal scale but it will also facilitate uncertainty modelling for specific system interactions. Importantly, this will allow for the analysis of multiple scenarios on the same diagram simultaneously which further contributes to the ability of the framework to account for system complexity. A CLD that illustrates quantitative feedback analyses will be easier to delimit in terms of size and what variables need to be included as the most important variables could be statistically identified in terms of the magnitude of impact on the final ES under consideration.

Bagstad *et al.* (2013) highlight how reducing resource and data requirements for ES models is essential to facilitate increased use in environmental and policy decision-making. Despite quantitative ES modelling being so data intensive, a fair portion of the data already exists in current environmental models. A combination of quantitative and qualitative modelling appears to be the optimal concession that addresses the primary limitations of either approach, but this can only be determined through further examination. Thus, there is potential for future investigation into the potential development of system dynamics-based software that specifically focuses on ESs and the approach developed in this study from both a qualitative and quantitative perspective. Alpizar *et al.* (2007) emphasised the importance of targeting as a technique of identifying the most important variables when analysing complex socio-ecological systems. Thus, future research should be aimed at identifying the most relevant and important process pathways and variables (within a specific context) and their specific data requirements for quantitative modelling while considering available resources. This could facilitate the development of semi-quantitative ES analysis models informed by qualitative information and quantitative data for the most relevant variables and process pathways, to the degree that the data is available (Busch *et al.*, 2012). Ultimately, an endeavour of this nature will attempt to address the primary issues around parameter estimation (limited data availability) and the low transparency of quantitative modelling, as well as the subjective nature of qualitative approaches.

Another research and/or application opportunity of the ESVCA framework could be to explore its potential to address the primary limitations of BBNs (outlined in section 2.5) as ES modelling and assessment tools. BBNs are restricted to using discrete variables and are highly limited in their ability to incorporate feedbacks (Landuyt *et al.*, 2013). Thus, incorporating these two approaches together would allow for the integration of quantitative and qualitative data to inform the model as well as incorporating various system feedbacks. More specifically, research would need to focus on the integration of both methods in a manner that addresses the limitations of each model for assessing and analysing ESs and ultimately conducting value chain analyses.

Another possible line of inquiry involves investigating the potential role that ESVCA models can play in addressing the conservation finance gap with a specific focus on holistic ecosystem conservation. Credit Suisse *et al.* (2014) describe the need for a movement away from donor funding towards public/private investment as well as the development of public private partnerships to address the need for conservation finance. This is directly associated with outcomes of the model presented in this study, specifically in terms of promoting private and public investment into the conservation of ecosystems through developing more inclusive value chains.

Further research could examine the commonalities between the specific outcomes of the approach presented in this study and those of ES accounting. The EC (2013) identifies three key areas of research, two of which could be assisted through the use of the ESVCA framework, namely: physical ecosystem accounting and communication and dissemination. Firstly, exploring the potential of ESVCA to guide physical ecosystem accounting in terms of differentiating between final and intermediate ESs for a specific beneficiary group. This could potentially prevent double counting, as the CLD would graphically illustrate all the relevant variables within the system. Secondly, there is potential to investigate the use of CLDs and system dynamics modelling as an alternative communication and illustration tool to assist environmental accounting.

Ferraro (2000) argues that popular ecosystem development initiatives that attempt to increase the provision of desired ESs are hindered in many ways and direct contracting initiatives would be a more effective solution. Such contracting approaches create artificial

markets through which individuals that provide ESs can benefit. A novel path of investigation would be to compare and contrast current conservation contracting programmes with the framework presented in this thesis in terms of their potential to conserve and promote ESs in developing nations. This could potentially lead to the development of a combined approach that uses both methods simultaneously to improve ecosystem functioning and further develop markets for ESs.

7.1.2. Practical Applications

Carpenter *et al.* (2009), Fisher *et al.* (2008) and Nel *et al.* (2011) describe the need for ES based research to focus on bridging the gap between science and policy development. Refraining from using the ESVCA framework in isolation is a step in the right direction as it has the potential to inform current management and policy decision-making frameworks, if applied effectively. Whether it is integrated into an existing framework or used alongside a decision-making process, more relevant information will be generated, from which policy and management can be advanced. Costanza and Ruth (1998) note the importance of placing transdisciplinary models within the general framework of 'adaptive management' (Holling, 1978) if they are to be effective. Therefore, such models and the corresponding policies and management decisions based upon them are not viewed as definitive, but rather as developing an adaptive research process; therefore, focusing on monitoring and feedback to extend model development and outcomes as opposed to using models to defend and obfuscate ineffective and inaccurate policy (Costanza and Ruth, 1998).

There is potential to use the ESVCA approach to assist in the use of the ARIES tool, outlined in section 2.5. As previously mentioned, the ARIES tool models the flow, production and use of ESs in a particular context. An ES based CLD can provide pivotal information about the dependencies of the various flows of ESs as well as their complex interlinkages, thus, facilitating the identification of key system variables and associated intermediate variables that need to be modelled to achieve a specific outcome. This approach could be used as a point of departure as to when to use the ARIES tool, as well as a tool to verify outcomes and logic flows of existing models. The ARIES tool requires extensive amounts of data to run and thus, developing an ES based CLD as a scoping technique could provide important insight into the system dynamics before the tool is actually run.

The integrative framework designed for context based decision-making in natural resources management, presented by Pollard *et al.* (2014) (section 2.5), could directly incorporate the ESVCA framework. The approach laid out by Pollard *et al.* (2014) incorporates a basic CLD development component with broad specialist engagement. Through incorporating the detailed ESVCA methodology into this framework, it could extend the basic scenario development approach to facilitate a more effective resilience analysis through the use of ES value chains. Importantly, the VSTEOP paradigm will be maintained throughout the process.

Another potential use of the approach developed in this study could be to inform the SEEA method (section 2.5) with regards to the links between anthropogenic activity and ESs. As the SEEA attempts to link all information to human activity in one way or another, the ESVCA approach could provide the information needed to understand the linkages between various environmental and anthropogenic intermediate and final ESs. All in all, it will provide a holistic system view which can be directly related to the values recorded for each service.

7.2 Conclusions

There is a global consensus amongst scientists and economists alike that the natural systems providing the necessary services that support and drive modern day society have entered a period of drastic change (Carpenter *et al.*, 2009; De Groot *et al.*, 2010b; COWI, 2014; Griggs *et al.*, 2013; MA, 2005; ONEMA, 2011; O'Neill *et al.*, 2010; Ostrom, 2009; Rockström *et al.*, 2009; Wallace, 2008; WRI, 2000). The concept of ESs is an extension of both ecological functioning and economic externalities that creates a nexus between the fields of ecology and economics, despite their seemingly conflicting goals (Fisher *et al.*, 2009). Global recognition of the economy as a sub-system of the environment and rigorous scientific research in ESs are in its infancy and thus, robust means for modelling, mapping, valuing and measuring ESs are non-existent and/or widely debated (Costanza, 2008; Fisher *et al.*, 2009; O'Neil *et al.*, 2010).

Ecosystems have experienced more anthropogenically induced change in the last 50 years than any other period in human history and these changes are only predicted to increase (Brauman *et al.*, 2007; Carpenter *et al.*, 2009; MA, 2005). Although these changes have the potential to improve the quantity and/or quality of the benefits received from ecosystems, the majority of change has had a negative impact on ecosystem functionality. In light of the

existing and predicted consequences of ecosystem degradation and the recent increase in scientific research surrounding ES delivery, modern society is undergoing a seminal change in the way benefits of natural ecosystems are understood and valued (Maltby and Acreman, 2011). Thus, further emphasising the importance of research focusing on providing practical solutions to the threats posed by diminishing provision of essential ESs.

The application of the ESVCA approach enabled the development of multiple AES based CLDs, which provide complex illustrations of the linkages between the various components of final AES value chains. The analysis of various scenarios allowed for the illustration of the ripple effects through individual parts of the value chain, which facilitated the identification of associated challenges and opportunities in specific value chains. Ultimately, this information can be used to recommend methods to improve AES provision and associated value chains. Effectively addressing all four research objectives identified in Chapter 1.

It is evident from the findings presented in this study that CLDs have the potential to facilitate an 'alternative' value chain analysis for AESs in the context of a catchment level case study. Despite not being able to conduct a value chain analysis in the traditional manner, the outcomes of the scenario analyses allowed for an 'alternative' value chain analysis to be completed that achieved the relevant goals of traditional value chain analyses. The ESVCA framework enables the identification of forward linkages and ripple effects in individual value chains of final AESs and the identification and assessment of challenges and opportunities in the value chains of final AESs and associated markets. Ultimately, facilitating the development of strategies and recommendations to improve the efficiency and effectiveness of final AES provision.

The ESVCA approach provides the framework through which progress towards understanding and integrating fully inclusive value chain analyses, which incorporate environmental processes and services, into policy and decision-making is a realistic outcome. The model is predictive in nature and thus, allows for a proactive approach towards ecosystem management that is geared towards improving the provision of chosen AESs while increasing system understanding for relevant stakeholders and decision-makers. Private and public entities that rely on the provision of ESs have the potential to take advantage of this approach to recognise potential opportunities and threats within the value chains of these ESs.

Identifying the most efficient methods of improving ES provision could significantly improve financial sustainability if taken advantage of. The relative accuracy of the model was validated through the outcomes of the scenario analyses which corroborates the reliability of the subsequent value chain analyses. Predictive, holistic and complex models of this nature are imperative if predicted future threats to global water supplies are to be mitigated and/or adapted to.

Global restoration of ESs and improvement of human well-being is a long-term, spatially and temporally complex experiment requiring constant learning and innovation. Incorporating different social, political and ecological dimensions is critical when investigating ESs within a specific decision context, as a consequence, trade-offs and conflict are inevitable. The development of multi-faceted models that present these trade-offs by incorporating different levels of complexity specific to the system can benefit equitable resource allocation. The ESVCA framework has the potential to contribute towards increasing the provision of essential AESs, improving human-welfare and the standard of living of the most vulnerable communities. However, these improvements may come along with environmental risks, thus, valuing, contrasting and comparing what is being traded off is vital for decision-making.

The ESVCA framework challenges fundamental economic ideologies surrounding the notion of infinite growth in a world with limited natural resources. It attempts to address central questions around complex socio-ecological systems while simultaneously assessing assumptions of policy and practice aimed at improving human well-being through improved provision of essential ESs. This is done by using the ES, CLD and value chain concepts to provide information to support long-term sustainable management of the socio-ecological systems upon which all life depends.

Ultimately, it is clear that CLDs have the potential to identify challenges and opportunities for potential and existing markets of AESs and thus, facilitate the development of recommendations for scenario adaption and/or mitigation, thereby improving AES value chains and their integration into formal markets. CLDs can thus be utilised as a suitable tool for an alternative approach towards value chain analyses for AESs, accomplishing the primary aim of this study. This method provides the framework for further research into ES value chains, sustainable management and complex system modelling.

In view of the future research potential of this method and various practical application opportunities, it is clear that multiple framework modelling and decision-making is the future of ES assessment, analysis and modelling. Combining complex models alongside multiple decision frameworks will provide the best opportunity for the information generated to be utilised as efficiently and effectively as possible and prevent research overlaps. Considering the potential uses and benefits associated with the conceptual approach established in this study, it is essential to continually develop the ESVCA framework to ensure policy and management decision-makers receive the most accurate, reliable and relevant information.

REFERENCE LIST



Photo: J.M. Rawlins

- AGUILERA, P.A., FERNANDEZ, A., RECHE, F. and RUMI, R. 2010. Hybrid Bayesian network classifiers: application to species distribution models. *Environmental Modelling & Software*, 25(12):1630-1639.
- ALPIZAR, F., BLACKMAN, A. and PFAFF, A. 2007. Payments for ecosystem services: Why precision and targeting matter. *Resources for the Future*, 165:20 -22.
- ARMAH, F.A., YAWSON, D.O., YENGOH, G.T., ODOI, J.O. and AFRIFA, E.K.A. 2010. Impact of floods on livelihoods and vulnerability of natural resource dependent communities in Northern Ghana. *Water*, 2:120-139.
- ARNOLD, J.G. and FOHRER, N. 2005. SWAT2000: current capabilities and research opportunities in applied watershed modeling. *Hydrological Processes*, 19(3):563-572.
- ARNOLD, J.G., SRINIVASIN, R., MUTTIAH, R.S. and WILLIAMS, J.R. 1998. Large area hydrologic modeling and assessment part I: model development. *Journal of the American Water Resources Association*, 34(1):73-89.
- ARQUITT, S. and JOHNSTONE, R. 2008. Use of system dynamics modelling in design of an environmental restoration banking institution. *Ecological Economics*, 65:63-75.
- BAGSTAD, K. J., SEMMENS, D. J., WAAGE, S. and WINTHROP, R. 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services*, e27-e39.
- BALETA, H. and PEGRAM, G. 2014. *Water as an input in the food value chain: Understanding the food energy water nexus*. WWF-SA, South Africa.
- BALMFORD, A., FISHER, B., GREEN, R.E., NAIDOO, R., STRASSBURG, B., TURNER R.K. and RODRIGUES, A.S.L. 2011. Bringing ecosystem services into the real world: An operational framework for assessing the economic consequences of losing wild nature. *Environmental and Resource Economics*, 48(2):161–175.
- BALMFORD, A., RODRIGUES, A., WALPOLE, M., TEN BRINK, P., KETTUNEN, M., BRAAT, L. and DE GROOT, R. 2008. *The Economics of Ecosystems and Biodiversity: Scoping the Science*. European Commission Report: ENV/070307/2007/486089/ETU/B2, Cambridge, UK.
- BARBIER, E.B. 2011. The policy challenges for green economy and sustainable economic development. *Natural Resources Forum*, 35:233-245.
- BARBIER, E.B. 2007. Valuing ecosystem services as productive inputs. *Economic Policy*, 49:178-229.
- BARTON, D. 2002. The transferability of benefits transfer. *Ecological Economics*, 42:147-164.

- BAUM, M.B. 2013. *Service Business Costing: cost accounting approach for the service industry*. Springer, Wiesbaden, Germany.
- BIGGS, H. and ROGERS, K.H. 2003. *An adaptive system to link science, monitoring and management in practice*. In: Du Toit, J.T., Rogers, K.H. and Biggs, H.C. (eds). *The Kruger experience: ecology and management of savanna heterogeneity*. Island Press: Washington, US.
- BISHOP, J., KAPILA, S., HICKS, F., MITCHELL, P. and VORHIES, F. 2008. *Building Biodiversity Business*. Shell International Limited and the International Union for Conservation of Nature, London, UK.
- BOLUND, P. and HUNHAMMAR, S. 1999, Ecosystem services in urban areas. *Ecological Economics*, 29(2):293-301.
- BOLWIG, S., PONTE, S., DU TOIT, A., RIISGAARD, L. and HALBERG, N. 2010. Integrating poverty and environmental concerns into value-chain analysis: a conceptual framework. *Development Policy Review*, 28(2):173-194.
- BOOYSEN, P.D. and TANTON, N.M. 1984. Ecological effects of fire in South African ecosystems. *Ecological Studies*, 48:1-18.
- BOSHOF, A.F. 2008. *The Baviaanskloof Mega-Reserve: from concept to implementation*. African Centre for Conservation Ecology Report: 58, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.
- BOSHOF, A.F. 2005. *The Baviaanskloof Mega-reserve: An environmentally, socially and economically sustainable conservation and development initiative*. Terrestrial Ecology Research Unit Report: 52, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.
- BOSHOF, A.F., COWLING, R.M. and KERLEY, G.I.H. 2000. *The Baviaanskloof Conservation Area – a conservation and tourism development priority*. Terrestrial Ecology Research Unit Report: 27, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.
- BOUMANS, R., ROMAN, J., ALTMAN, I. and KAUFMAN, L. 2015. The Multiscale Integrated Model of Ecosystem Services (MIMES): simulating the interactions of coupled human and natural systems. *Ecosystem Services*, 12:30-41.
- BOUMANS, R., COSTANZA, R., FARLEY, J., WILSON, M.A., PORTELA, R., ROTMANS, J., VILLA, F. and GRASSO, M. 2002. Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model. *Ecological Economics*, 41:529- 560.

- BOYD, J. and BANZHAF, S. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, 63:616-626.
- BOYD, 2007. Nonmarket benefits of nature: what should be counted in green GDP. *Ecological Economics*, 61(4):716-723.
- 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):4-15.
- BRAUMAN, K.A., DAILY, G.C., DUARTE, T.K. and MOONEY, H.A. 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. *Annual Review of Environment and Resources*, 32:6.1-6.32.
- BRAUN, V. and CLARKE, V. 2013. *Successful qualitative research: a practical guide for beginners*. SAGE Publications, Los Angeles, US.
- BRITTON, T., BROUWERS, L. and KUHLMANN. 2012. *System Analyses 1*. Stockholm, Sweden. [Online]. Available: http://kurser.math.su.se/moodle19/file.php/793/SD2_Stock_and_Flows_Sys_Dyn_text.pdf. [25/05/2015].
- BROWN, T.C., BERGSTROM, J.C. AND LOOMIS, J.B. 2007 Defining, valuing, and providing ecosystem goods and services. *Natural Resources Journal*, 47(2):329–376.
- BUNN, S.E. and ARTHINGTON, A.H. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management*, 30(4):492-507.
- CARDOSO, A.C., EGOH, B., FIORINA, C., ZAL, N., KRISTENSEN, P., WERNER, B., ERHARD, M., VANNEUVILLE, W., PIETO, O.G. and ROMANIWIC, A. 2013. *Working paper: Tailoring of the Common framework for ecosystem assessment for freshwater ecosystems*. MAES Pilot 4. Joint Research Centre, European Commission, Luxembourg City, Luxembourg.
- CARPENTER, S.R., MOONEY, H.A., AGARD, J., CAPISTRANO, D., DEFRIES, R.S., DIAZ, S., DIETZ, T., DURAIAPPAH, A.K., OTENG-YEBOAH, A., PEREIRA, H.M., PERRINGS, C., REID, W.V., SARUKHAN, J., SCHOLE, R.J. and WHYTE, A. 2009. Science for managing ecosystem services: beyond the Millennium Ecosystem Assessment. *PNAS*, 106(5):1305-1312.
- CARPENTER, S. R., E. M. BENNETT, AND G. D. PETERSON. 2006. Scenarios for ecosystem services: an overview. *Ecology and Society*, 11(1):29.
- CARVALHO-SANTOS, C., HONRADO, J.P. and HEIN, L. 2014. Hydrological services and the role of forests: conceptualization and indicator-based analysis with an illustration at a regional scale. *Ecological Complexity*, 20:69-80.

- UN (United Nations). 2008. *Biodiversity and Agriculture: Ecosystem services for human well-being*. United Nations Convention on Biological Diversity, New York. [Online]. Available: <https://www.cbd.int/doc/bioday/2008/ibd-2008-factsheet-01-en.pdf>. [13/10/2015].
- UN. 1992. *Convention on Biological Diversity*. United Nations Convention on Biological Diversity. United Nations, New York. [Online]. Available: <https://www.cbd.int/doc/legal/cbd-en.pdf>. [13/10/2015].
- CHAN, K.M.A., SHAW, M.R., CAMERON, D.R., UNDERWOOD, E.C. and DAILY, G.C. 2006. Conservation planning for ecosystem services. *PLOS Biology*, 4(11):2138-2152.
- CHRISTIE, M., HANLEY, N., WARREN, J., MURPHY, K., WRIGHT, R. and HYDE, T. 2006. Valuing the diversity of biodiversity. *Ecological Economics*, 58:304-317.
- CHRISTENSEN, V., STEENBEEK, J. and FAILLER, P. 2011. A combined ecosystem and value chain modeling approach for evaluating societal cost and benefit of fishing. *Ecological Modelling*, 222(3):857-864.
- CILLIERS, S., CILLIERS, J., LUBBE, R. and SIEBERT, S. 2012. Ecosystem services of urban green spaces in African countries – perspectives and challenges. *Urban Ecosystems*, 16(4): 681-702.
- COSTANZA, R., DE GROOT, R., SUTTON, P., VAN DER PLOEG, S., ANDERSON, S.J., KUBISZEWSKI, I., FARBER, S. and TURNER, R.K. 2014. Changes in the global value of ecosystem services. *Global Environmental Change*, 26:152-158.
- COSTANZA, R. 2008. Ecosystem services: Multiple classification systems are needed. *Biological Conservation*, 141:350-352.
- COSTANZA, R. and RUTH, M. 1998. Using dynamic modelling to scope environmental problems and build consensus. *Environmental Management*, 22(2):183-195.
- COSTANZA, R., D'ARGE, R., DE GROOT, R., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O'NEILL, R.V., PARUELO, J., RASKIN, R.G., SUTTON, P. and VAN DEN BELT, M. 1998. The value of ecosystem services: putting the issues in perspective. *Ecological Economics*, 25:67-72.
- COSTANZA, R., D'ARGE, R., DE GROOT, R., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O'NEILL, R.V., PARUELO, J., RASKIN, R.G., SUTTON, P. and VAN DEN BELT, M. 1997. The value of the World's ecosystem services and natural capital. *Nature*, 387:253-260.

- COWI. 2014. *Support policy development for integration of ecosystem services assessments into WFD and FD implementation*. Annex Report. [Online]. Available: https://circabc.europa.eu/sd/a/79584b30a2a34fa89d0785303d9b1/Ecosystem%20services_WFD_FD_Annex%20report_Final.pdf. [01/06/2015].
- CREDIT SUISSE, WWF and MCKINSEY & COMPANY. 2014. *Conservation finance: Moving beyond donor funding to an investor driven approach*. Available: https://unfccc.int/files/cooperation_and_support/financial_mechanism/standing_committee/application/pdf/conservation_finance_-_moving_beyond_donor_funding_toward_an_investor-driven_approach.pdf. [10/12/2015].
- CRESSWELL, J.W. 2013. *Research design: qualitative, quantitative, and mixed methods approaches* (4e). SAGE Publications, Los Angeles, US.
- CROCKER, T. D. and J. TSCHIRHART. 1992. Ecosystems, externalities and economics. *Environment and Resource Economics*, 2:551-567.
- DAILY, G.C., POLASKY, S., GOLDSTEIN, J., KAREIVA, P.M., MOONEY, H.A., PEJCHAR, L., RICKETTS, T.H., SALZMAN, J. and SHALLENBERGER, R. 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment*, 7(1):21-28.
- DAILY, G. C. 1997. *Nature's services: Societal dependence on natural ecosystems*. Island Press, Washington, US.
- DAWNAY, E. and SHAH, H. 2005. *Behavioural economics: Seven principles for policy-makers*. New Economics Foundation publications, London, UK.
- DE GROOT, R.S., FISHER, B., CHRISTIE, M., ARONSON, J., BRAAT, L., GOWDY, J., HAINES-YOUNG, R., MALTBY, E., NEUVILLE, A., POLASKY, S., PORTELA, R. and RING, I. 2010a. *Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation*. In: Kumar, P. (ed). *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. Earthscan, London, UK.
- DE GROOT, R.S., ALKEMADE, R., BRAAT, L., HEIN, L. and WILLEMEN, L. 2010b. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, 7:260-272.
- DE GROOT, R.S., WILSON, M.A. and BOUMANS, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41:393-408.
- DIXON, J.A. and PAGIOLA, S. 1998. *Economic analysis and environmental assessment*. Environmental assessment source book update. World Bank, Washington, US.

- DOYLE, J.K. and FORD, D.N. 2000. Mental models concepts revisited: some clarifications and a reply to Lane. *System Dynamics Review*, 15:411–415.
- DOYLE, J.K. and FORD, D.N. 1998. Mental models concepts for system dynamics research. *System Dynamics Review*, 14:3–29.
- DWA (Department of Water Affairs). 2013a. *National Water Resources Strategy: water for an equitable and sustainable future* (Second Edition). Department of Water Affairs, Pretoria, South Africa.
- DWA. 2013b. *National Water Policy Review: Updated policy positions to overcome the water challenges of our developmental state to provide for improved access to water, equity and sustainability*. Government Gazette: 36798. Department of Water Affairs, Pretoria, South Africa.
- DWAF (Department of Water Affairs and Forestry). 2003. *Fish to Tsitsikamma Water Management Area: Overview of Water Resource Availability and Utilisation*. Department of Water Affairs and Forestry Report: P 15/000/00/0203, Pretoria, South Africa.
- DWAF. 2002. *Fish to Tsitsikamma Water Management Area: Water Resources Situation Assessment*. Department of Water Affairs and Forestry Report: P 15000/00/0101, Volume 2, Pretoria, South Africa.
- DWS (Department of Water and Sanitation). 2014. *Resource Quality Services*. Department of Water and Sanitation. [Online]. Pretoria, South Africa. Available: <http://www.dwaf.gov.za/iwqs/default.aspx>. [09/02/2015].
- EC (European Commission). 2013. *System of Environmental-Economic Accounting*. European Commission, Organisation for Economic Co-operation and Development, United Nations, World Bank [Online]. Available: http://unstats.un.org/unsd/envaccounting/eea_white_cover.pdf. [29/05/2015].
- EEA (European Environment Agency). 2015. *CICES: Towards a common classification of ecosystem services*. European Environment Agency. [Online]. Available: <http://cices.eu/>. [01/06/2015].
- EGOH, B., REYERS, B., ROUGET, M., RICHARDSON, D.M., LE MAITRE, D.C. and VAN JAARSVELD, A.S. 2008. Mapping ecosystem services for planning and management. *Agriculture, Ecosystems and Environment*, 127(1-2):135-140.
- ELLIS, G.M. and FISHER, A.C. 1987. Valuing the environment as input. *Journal of Environmental Management*, 25:149-56.
- EVANS, D. 2004. *A comparative valuation of ecosystem services: Lents project case study*. Prepared for city of Portland watershed management program, ECONorthwest.

- FARBER, S.C., COSTANZA, R. and WILSON, M.A. 2002. Economic and ecological concepts for valuing ecosystem services. *Ecological Economics*, 41:375-392.
- FELLER, A., SHUNK, D. and CALLARMAN, T. 2006. *Value Chains Versus Supply Chains*. [Online]. BPTrends. Available: <http://www.ceibs.edu/knowledge/papers/images/20060317/2847.pdf>. [13/03/2015].
- FERRARO, P.J. 2000. *Constructing Markets for Ecosystem Services: limitations of development interventions and a role for conservation performance payments*. 8th International Association for the Study of Common Property Conference, Indiana University, Bloomington, Indiana, US.
- FINNOFF, D., SHOGREN, J.F., LEUNG, B. and LODGE, D. 2005. The importance of bioeconomic feedback in invasive species management. *Ecological Economics*, 52:367-381.
- FISCHER, A. and HANLEY, N. 2007. Analysing decision behaviour in stated preference surveys: A consumer psychological approach. *Ecological Economics*, 61:303-314.
- FISHER, B., TURNER, R.K. and MORLING, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics*, 68:643-653.
- FISHER, B. and TURNER, K. 2008 Ecosystem services: Classification for valuation. *Biological Conservation*, 141(5):1167–1169.
- FISHER, B., TURNER, K., ZYLSTRA, M., BROUWER, R., DE GROOT, R., FARBER, S., FERRARO, P., GREEN, R., HADLEY, D., HARLOW, J., JEFFERISS, P., KIRKBY, C., MORLING, P., MOWATT, S., NAIDOO, R., PAAVOLA, J., STRASSBURG, B., YU, D. and BALMFORD, A. 2008. Ecosystem services and economic theory: integration for policy-relevant research. *Ecological Applications*, 18(8):2050–2067.
- FERIS, L.A. 2010. The role of good environmental governance in the sustainable development of South Africa. *Potchefstroom Electronic Law Journal*, 13(1):72-99.
- FERNALD, A., TIDWELL, V., RIVERA, J., RODRIGUEZ, S., GULDAN, S., STEELE, C., OCHOA, C., HURD, B., ORTIZ, M., BOYKIN, K. and CIBILS, A. 2012. Modeling sustainability of water, environment, livelihood, and culture in traditional irrigation communities and their linked watersheds. *Sustainability*, 4:2998-3022.
- FORD, A. 1999. *Modeling the Environment: An Introduction to System Dynamics Modeling of Environmental Systems*. Island Press, Washington, US.
- FORRESTER, J.W. 1994. System dynamics, systems thinking, and soft OR. *System Dynamics Review*, 10:245–256.
- FORRESTER, J.W. 1975. *Collected Papers of Jay W. Forrester*. Wright-Allen Press, Cambridge, UK.

- FORRESTER, J.W. 1968. *Principles of Systems*. Pegasus Communications, Waltham, US.
- FORRESTER, J. W. 1961. *Industrial Dynamics*. Pegasus Communications, Waltham, US.
- FOURRETURNS. 2014. *The FourReturns Development Company*. Available: <http://www.fourreturns.co.za/>. [05/09/2015].
- GALLOPIN, G. 2003. *A systems approach to sustainability and sustainable development*. United Nations Publication: LC/L 1864-P, New York, US.
- GARROD, G., and WILLIS, K.G. 1999. *Economic Valuation of the environment*. Edward Elgar, Cheltenham, UK.
- GEREFFI, G., HUMPHREY, J., KAPLINSKY, R. and STURGEON, T. 2001. Globalisation, value chains and development. *IDS Bulletin*, 32(3):1-12.
- GILG, A. and BARR, S. 2006. Behavioural attitudes towards water saving? Evidence from a study of environmental actions. *Ecological Economics*, 57:400-414.
- 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*. Water Research Commission Report: TT 462/10, Pretoria, South Africa.
- GLENDAY, J. 2015. Personal communication. LivingLands research assistant. Rhodes University, Grahamstown, South Africa.
- 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):1209-1218.
- GOODSTEIN, E. S. 2011. *Economics and the environment* (6e). John Wiley & Sons, Portland, US.
- GRIGGS, D., STAFFORD-SMITH, M., OWEN, G., ROCKSTRÖM, J., ÖHMAN, M.C., SHYAMSUNDAR, P., STEFFEN, W., GLASER, G., KANIE, N. and NOBLE, I. 2013. Sustainable development goals for people and planet. *Nature*, 495:305-307.
- GRIGG, A., CULLEN, Z., FOXALL, J., CROSBIE, L., JAMISON, L. and BRITO, R. 2009. *The Ecosystem Services Benchmark*. A guidance document. Fauna & Flora International, United Nations Environment Programme Finance Initiative and Fundação Getulio Vargas.
- GRIZZETTI, B., LANZANOVA, D., LIQUETE, C. and REYNAUD, A. 2015. *Cook-book for water ecosystem service assessment and valuation*. Institute for Environment and Sustainability, Joint Research Centre, European Commission. European Commission Report: EUR 27141 EN.

- GRÖSSER, S.N. 2012. *Co-evolution of standards in innovation systems: the dynamics of voluntary and legal building codes*. Springer Science and Business Media, Heidelberg, Germany.
- GULL, K. 2009. *The economics of a payment for watershed services in the western Baviaanskloof*. Unpublished Honours Thesis. Rhodes University, Grahamstown, South Africa.
- GUO, H.C., LIU, L., HUANG, G.H., FULLER, G.A., ZOU, R. and YIN, Y.Y. 2001. A system dynamics approach for regional environmental planning and management: a study for the Lake Erhai Basin. *Journal of Environmental Management*, 61:93-111.
- GUSWA, A.J., BRAUMAN, K.A., BROWN, C., HAMEL, P., KEELER, B.L. and SAYRE, S. 2014. Ecosystem services: challenges and opportunities for hydrologic modelling to support decision making. *Water Resources Research*, 50:4535-4544.
- HAINES-YOUNG, R. and POTSCHIN, M. 2014. *Typology/Classification of Ecosystem Services*. In: Potschin, M. and K. Jax. (eds). OpenNESS Ecosystem Services Reference Book. EC FP7 Grant Agreement: 308428.
- HAINES-YOUNG, R. and POTSCHIN, M. 2013. *Common International Classification of Ecosystem Services (CICES): Consultation version 4, August-December 2012*. EEA Framework Contract: EEA/IEA/09/003.
- HAINES-YOUNG, R. and POTSCHIN, M. 2011. *Common International Classification of Ecosystem Services (CICES) 2011 Update*. Paper prepared for the UN/World Bank/EEA Expert meeting on Ecosystem Accounts, London, UK. EEA Framework Contract: EEA/BSS/07/007.
- HALE, E.O. 1993. Successful public involvement. *Journal of Environmental Health*, 55(4):17-19.
- HANLEY, N., SHOGREN, J.F. and WHITE, B. 2007. *Environmental economics in theory and practice*. Palgrave Macmillan, Beijing, China.
- HARTWICK, J.M. 1978. Investing returns from depleting renewable resource stocks and intergenerational equity. *Economic letters*, 1(1):85-8.
- HEAL, G. 2000. *Nature and the Marketplace: Capturing the Value of Ecosystem Services*. Island Press, Washington, US.
- 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):209-228.

- HENDERSON, J., DICKEN, P., HESS, M., COE, N. and YEUNG, H.W. 2002. Global production networks and the analysis of economic development. *Review of International Political Economy*, 9(3):436-464.
- HEPBURN, C., DUNCAN, S. and PAPACHRISTODOULOU, A. 2010. Behavioural economics, hyperbolic discounting and environmental policy. *Environmental Resource Economics*, 46:189-206.
- HERGERT, M. and MORRIS, D. 1989. Accounting data for value chain analysis. *Strategic Management Journal*, 10:175-188.
- HITCHENS, D.K. 2005. *Systems Thinking*. [Online]. Available: <http://www.hitchins.net/systems/systems-thinking/systems-thinking.html>. [25/05/2015].
- HJORTH, P. and BAGHERI, A. 2006. Navigating towards sustainable development: a systems dynamics approach. *Futures*, 38:74-92.
- HOLLING, C.S. 1978. *Adaptive environmental assessment and management*. John Wiley and Sons, Chichester, UK.
- HOUGHTON, J. 2004. *Global Warming: The Complete Briefing* (3e). Cambridge University Press, Cambridge, UK.
- HUMPHREY, J. and H. SCHMITZ. 2001. Governance in Global Value Chains. *IDS Bulletin*, 32(3):17.
- HÜRLIMANN, M. 2009. *Dealing with real-world complexity: Limits, enhancements and new approaches for policy makers*. Gabler, Frankfurt, Germany.
- ILLGNER, P.M. AND HAIGH, E.H. 2003. *Identification, mapping and present state assessment of the wetlands in the Baviaanskloof River Catchment*. Report for Working for Wetlands. Eastern Cape IWR Wetland Group. Institute for Water Research, Rhodes University, Grahamstown, South Africa.
- IMA (Institute of Management Accountants). 1996. *Value chain analysis for assessing competitive advantage*. Statements on Management Accounting. Available: http://www.imanet.org/docs/default-source/thought_leadership/management_control_systems/value_chain_analysis_for_assessing_competitive_advantage.pdf?sfvrsn=2. [14/09/2015].
- IPCC (Intergovernmental Panel on Climate Change). 2013. *Climate Change 2013: the Physical Science Basis*. World Health Organisation, United Nations Environment Programme. [Online]. Available: <http://www.climatechange2013.org/report/full-report/>. [14/05/2015].

- IRELAND, R.D., HOSKISSON, R.E. and HITT, M.A. 2009. *The management of strategy*. Cengage, Mason, US.
- IUCN (International Union for the Conservation of Nature and Natural Resources). 1980. *World Conservation Strategy: Living Resource Conservation for Sustainable Development*. United Nations Environment Programme, World Wildlife Fund. [Online]. Available: <https://portals.iucn.org/library/efiles/edocs/WCS-004.pdf>. [16/04/2015].
- JACK, B.K., KOUSKY, C. and SIMS, K.R.E. 2008. Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. *PNAS*, 105(28):9465-9470.
- JAFARI, M., AMIRI, R.H. and BOUROUTI, A. 2008. *An interpretive approach to drawing causal loop diagrams*. Proceedings of the 26th International Conference of the System Dynamics Society, July 2008, Athens, Greece.
- JANSEN, H.C. 2008. *Water for food and ecosystems in the Baviaanskloof Mega Reserve: land and water resources assessment in the Baviaanskloof, Eastern Cape Province, South Africa*. Alterra Report: 1812, Wageningen University, Wageningen, Netherlands.
- JAX, K. 2010. *Ecosystem Functioning*. Cambridge University Press, Cambridge, UK.
- JAX, K. 2005. Function and 'functioning' in ecology: what does it mean? *OIKOS*, 111(3):641-648.
- JENSEN, F.V. 2001. *Bayesian Networks and Decision Graphs*. Springer, New York, US.
- JOHNSTON, R.J. and RUSSELL, M. 2011. An operational structure for clarity in ecosystem service values. *Ecological Economics*, 70:2243-2249.
- JORDAAN, H. and GROVÉ, B. 2012. *An economic analysis of the contribution of water use to value chains in agriculture*. Water Research Commission Report: 1779/1/12, Pretoria, South Africa.
- KAHNEMAN, D. 2003a. A perspective on judgment and choice: Mapping bounded rationality. *American Psychologist*, 58:697-720.
- KAHNEMAN, D. 2003b. A psychological perspective on economics. *American Economic Review*, 93:162-168.
- KAHNEMAN, D. 1986. *The review panel assessment: comment*. In: Cummings, R.G., Brookshire, D.S. and Schulze, W.D. (eds.). *Valuing Public Goods: The Contingent Valuation Method*. Rowman and Allanheld, Totowa, US.

- KAPLINSKY, R. and MORRIS, M. 2000. *Handbook for value chain research*. Prepared for the International Development Research Centre. [Online]. Available: <https://www.ids.ac.uk/ids/global/pdfs/VchNov01.pdf>. [17/10/2014].
- KAPLINSKY, R. 2004. Spreading the gains from globalisation: What can be learned from value chain analysis? *Problems of Economic Transition*, 47(2):74-115.
- KELLY, K.L. 1998. A systems approach to identifying decisive information for sustainable development. *European Journal of Operational Research*, 109(2):452-464.
- KETCHEN, D.J. and HULT, G.T.M. 2007. Bridging organization theory and supply chain management: the case of the best value supply chains. *Journal of Operations Management*, 25(2):573-580.
- KIRKMAN, K.E. 2006. *Baviaans conservancy feasibility study: feasibility study to investigate the conversion of land use from domestic stock farming to sustainable biodiversity-based ventures*. Unpublished Report. C.A.P.E. programme and Terrestrial Ecology Research Unit, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.
- KIRKWOOD, C.W. 2013. *Chapter 1: System behaviour and causal loop diagrams*. Arizona State University, Arizona, US. [Online]. Available: <http://www.public.asu.edu/~kirkwood/sysdyn/SDIntro/ch-1.pdf>. [05/05/2015].
- KNIGHT, F.H. 2012. *Agricultural assessment of the Baviaanskloof*. Report prepared for Greenchoice. Agri Informatics Development Trust, Durbanville, South Africa.
- KOCA, D. and SVERDRUP, H. 2012. *Use of causal loop diagrams and systems analysis to explore alternative climate change adaption strategies in Seyhan River Basin, Turkey*. Proceedings of the 30th International Conference of the System Dynamics Society, July 2012, Gallen, Switzerland.
- KOTZE, D., MARNEWECK, G., BATCHELOR, A., LINDLEY, D. and COLLINS, N. 2008. *WET-EcoServices: A technique for rapidly assessing ecosystem services supplied by wetlands*. Water Research Commission Report: TT 339/09, Pretoria, South Africa.
- KRUGER, P. 2015. Personal communication. Baviaanskloof farmer. Zandvlakte farm, Baviaanskloof, South Africa.
- LANDERS, D.H. AND NAHLIK, A.M. 2013. *Final Ecosystem Goods and Services Classification System (FEGS-CS)*. Environmental Protection Agency Report: EPA/600/R-13/ORD-004914, Washington, USA.

- LANDUYT, D., LEMMENS, P., D'HONDT, R., BROEKX, S., LIEKENS, I., DE BIE, T., DECLERCK, S.A.J., DE MEESTER, L. and GOETHALS, P.L.M. 2014. An ecosystem service approach to support integrated pond management: a case study using Bayesian belief networks – highlighting opportunities and risks. *Environmental Management*, 145:79-87.
- LANDUYT, D., BROEKX, S., D'HONDT, R., ENGELEN, G., AERTSENS, J. and GOETHALS, P.L.M. 2013. A review of Bayesian belief networks in ecosystem service modelling. *Environmental Modelling & Software*, 46:1-11.
- LANE, D.C. 2008. The Emergence and use of diagramming in system dynamics: a critical account. *Systems Research and Behavioral Science*, 25:3-23.
- LANE, D.C. 1999. Friendly amendment: a commentary on Doyle and Ford's proposed re-definition of 'mental model'. *System Dynamics Review*, 15:185– 194.
- LANEN, W.N., ANDERSON, S.W. and MAHER, M.W. 2008. *Fundamentals of cost accounting*. McGraw-Hill/Irwin, New York, US.
- LARKIN, J.H. and SIMON, H.A. 1987. Why a diagram is (sometimes) worth ten thousand words. *Cognitive Science*, 11:65–99.
- LAW, J. 1999. *After ANT: complexity, naming and typology*. In: LAW, J. and Hassard, J. (eds). Actor-network Theory and After. Blackwell, Oxford, UK.
- LEIBEL, N. 2012. *Protecting biodiversity in production landscapes : a guide to working with agribusiness supply chains towards conserving biodiversity*. United Nations Development Programme, New York, US.
- LI, Z. and ZHANG, J. 2001. Calculation of field Manning's Roughness Coefficient. *Agricultural Water Management*, 49(2):153-161.
- LIANG, X., LETTENMAIER, D.P., WOOD, E.F. and BURGESS, S.J. 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. *Journal of Geophysical Research*, 99(7):14415-14428.
- LIMBURG, K.E. 2009. Aquatic ecosystem services. *Encyclopedia of Inland Waters*, 1:25-30.
- LIMBURG, K.E., O'NEILL, R.V., COSTANZA, R. and FARBER, S. 2002. Complex systems and valuation. *Ecological Economics*, 41:409-420.
- LIMERINOS, J.T. 1970. *Determination of the Manning coefficient from measured bed roughness in natural channels*. US Geological Survey, Water Supply Paper: 1898-B, Wisconsin, US.
- LOEHE, C. 2004. Challenges of ecological complexity. *Ecological Complexity*, 1:3–6.

- LYNCH, R. 2003. *Corporate strategy*. Financial Times/Prentice Hall, Essex, UK.
- MA (Millennium Ecosystem Assessment). 2005. *Ecosystems and human well-being: wetlands and water synthesis*. World Resources Institute, Washington, DC.
- MAES (Mapping and Assessment of Ecosystem and their Services). 2013. *An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020*. European Commission Technical Report: 067.
- MALTBY, E. and ACREMAN, M. 2011. Ecosystem services of wetlands: pathfinder for a new paradigm. *Hydrological Sciences Journal*, 56(8):1341-1359.
- MANDER, M., BLIGNAUT, J., VAN NIEKERK, M.A., COWLING, R., HORAN, M.J.C., KNOESEN, D.M., MILLS, A., POWELL, M. and SCHULZE, R.E. 2010. *Baviaanskloof – Tsitsikamma Payment for ecosystem services: A feasibility assessment*. South African National Biodiversity Institute, Pretoria, South Africa.
- MANUEL-NAVARRETE, D., GOMEZ, J.J. and GALLOPIN, G. 2007. Syndromes of sustainability of development for assessing the vulnerability of coupled human-environmental systems. The case of hydrometeorological disasters in Central America and the Caribbean. *Global Environmental Change*, 17:207-217.
- MARTINEZ-ALIER, J. 2005. *Ecological economics as human ecology*. In: Sahu, N. and Choudjary, A. (eds). *Dimensions of Environmental and Ecological Economics*. Universities Press, Hyderabad, India.
- MCCOMB, G., LANTZ, V., NASH, K. and RITTMASER, R. 2006. International valuation databases: overview, methods and operational issues. *Ecological Economics*, 60:461-472.
- MCCUTCHEON, D. and MEREDITH, J. 1993. Conducting case study research in operations management. *Journal of Operations Management*, 11(3):239-56.
- MENDELSON, S. 2015. Personal communication. LivingLands Research Coordinator. PRESENCE Learning Village, Eastern Cape, South Africa.
- MUETZELFELDT, R. and MASSHEDER, J. 2003. The Simile visual modelling environment. *European Journal of Agronomy*, 18:345–358.
- MIDGLEY, S.J.E., CHESTERMAN, S. and HOPE, E. 2012. *Payment for ecosystem services: a climate change adaptation strategy for southern Africa*. Regional Climate Change Programme for Southern Africa (RCCP), UK Department for International Development (DFID). OneWorld Sustainable Investments, Cape Town, South Africa.

- MINDTOOLS. 2014. *Value Chain Analysis: Achieving Excellence in Things that Really Matter*. [Online]. Available: www.mindtools.com/pages/articles/newTMC_10.htm. [12/09/2015].
- MOFFATT, I. and HANLEY, N. 2001. Modelling sustainable development: systems dynamic and input-output approaches. *Environmental Modelling & Software*, 16:545-557.
- MORECROFT, J.D.W. and STERMAN, J.D. 1994. *Modeling for Learning Organizations*. Productivity Press, Portland, US.
- MORECROFT, J.D.W. 1982. A critical review of diagramming tools for conceptualising feedback system models. *Dynamica*, 8:20–29.
- MOWEN, M. M. and HANSEN, D.R. 2011. *Introduction to cost accounting*. Cengage, Spiceland, US.
- MOXNES, E. 2004. Misperceptions of basic dynamics: the case of renewable resource management. *System Dynamics Review*, 20:139–162.
- MULLAINATHAN, S. and THALER, R. H. 2000. *Behavioural economics*. Massachusetts Institute of Technology Working Paper: 00-27, Department of Economics. Cambridge, Massachusetts, US.
- MÜLLER, F. and BURKHARD, B. 2012. The indicator side of ecosystem services. *Ecosystem Services*, 1(1):26-30.
- MURDOCH, J. 1998. The spaces of actor-network theory. *Geoforum*, 29(4):357–74.
- NAHMAN, A., DE LANGE, W., WISE, R. and PITSO, D. 2009. *Environmental and resource economics in South Africa: A review*. Council for Scientific and Industrial Research Report: CSIR/NRE/RBSD/IR/2007/0092/A, Stellenbosch, South Africa.
- NCP (Natural Capital Project). 2016. *InVEST: integrated valuation of ecosystem services and tradeoffs*. [Online]. Available: <http://www.naturalcapitalproject.org/invest/#what-is-invest>. [09/11/2015].
- NEL, J., COLVIN, C., LE MAITRE, D., SMITH, J. and HAINES, I. 2013. *South Africa's Strategic Water Source Areas*. Council for Scientific and Industrial Research Report: CSIR/NRE/ECOS/ER/2013/0031/A, Pretoria, South Africa.
- NEL, J., DRIVER, A., STRYDOM, W.F., MAHERRY, A., PETERSEN, C., HILL, L., ROUX, D.J., NIENABER, S., VAN DEVENTER, H., SWARTZ, E. and SMITH-ADAO, L.B. 2011. *Atlas of Freshwater Ecosystem Priority Areas in South Africa: Maps to support sustainable development of water resources*. Water Research Commission Report: TT 500/11, Pretoria, South Africa.

- NEL, J.L., SMITH-ADAO, L., ROUX, D.J., ADAMS, J., CAMBRAY, J.A., DE MOOR, F.C., KLEYNHANS, C.J., KOTZE, I., MAREE, G., MOOLMAN, J., SCHONEGEVEL, L.Y., SMITH, R.J. and THIRION, C. 2006. *Conservation Planning for River and Estuarine Biodiversity in the Fish-to-Tsitsikamma Water Management Area*. Water Research Commission Report: K5/1486, Pretoria, South Africa.
- NIJSEN, B.N., LETTENMAIER, D.P., LIANG, X., WETZEL, S.W. and WOOD, E.F. 1997. Streamflow simulation for continental-scale river basins. *Water Resources Research*, 33(4):711-724.
- NOIRTIN, E., 2008. *Analysis of stakeholder interactions and involvement in thicket restoration, Western Baviaanskloof, South Africa Who is willing to restore and why?* Unpublished MSc thesis, Department of Environmental Sciences, Wageningen University, Wageningen, Netherlands.
- NPC (National Planning Commission). 2012. *National Development Plan 2030: Our Future – Make it work*. National Planning Commission, South Africa.
- NUNES, P.A.L.D., KUMAR, P. and DEDEURWAERDERE, T. 2014. *Handbook on the economics of ecosystem services and biodiversity*. Edward Elgar: Northampton.
- ONEMA (The National Agency for Water and Aquatic Environments). 2011. *Assessing the ecological services of aquatic environments scientific, political and operational issues*. [Online]. Available: <http://www.onema.fr/IMG/EV/publication/ecological-services/ecological-services-couv.pdf>. [12/05/2015].
- O'NEILL, D.W., DIETZ, R. and JONES, N. 2010. *Enough is Enough: Ideas for a sustainable economy in a world of finite resources*. The report of the Steady State Economy Conference. Center for the Advancement of the Steady State Economy and Economic Justice for All, Leeds, UK.
- OSTROM, E. 2009. A general framework for analysing sustainability of social-ecological systems. *Science*, 325(1):419-422.
- OSTROM, E. 2007. A diagnostic approach for going beyond panaceas. *PNAS*, 104(39):15181-15187.
- PALMER, M.A., FILOSO, S. and FANELLI, R.M. 2014. From ecosystems to ecosystem services: stream restoration as ecological engineering. *Ecological Engineering*, 65:62-70.
- PALMER, C.G., BEROLD, R.S. AND MULLER, W.J. 2004. *Environmental water quality in water resources management*. Water Research Commission Report: TT 217/04, Pretoria, South Africa.
- PARKS, S. and GOWDY, J. 2013. What have economists learned about valuing nature? A review essay. *Ecosystem Services*, 3:1-10.

- PASCUAL, U., MURADIAN, R., BRANDER, L., GOMEZ-BAGGETHUN, E., MARTIN-LOPEZ, B. and VERMA, M. 2010. *The economics of valuing ecosystem services and biodiversity*. In: Kumar, P. (ed). *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. Earthscan, London and Washington, UK and US.
- PATTON, M.Q. 2001. *Qualitative research and evaluation methods*. SAGE Publications, Los Angeles, US.
- PEPPARD, J. and RYLANDER, A. 2006. From value chain to value network: insights for mobile operators. *European Management Journal*, 24(2-3):128-141.
- PERMAN, R., MA, Y. and MCGILVRAY, J. 1996. *Natural Resource and Environmental Economics*. Longman: London and New York.
- PIRARD, R., BILLE, R. and SEMBRES, T. 2010. *Questioning the theory of Payments for Ecosystem Services (PES) in light of emerging experience and plausible developments*. Institute for Sustainable Development and International Relations, Paris, France.
- PLUMMER, M.L. 2009. Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment*, 7:38–45.
- POLLARD, S., BIGGS, H. and DU TOIT, D.R. 2014. A systematic 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):63-80.
- POLLARD, S.R., RYDANNYKH, A., DU TOIT, D.R. and LAPORTE-BISQUIT, A. 2013. *Development of a participatory framework for understanding water-related ecosystem services within the context of classification and the reserve*. Water Research Commission Report: TT 574/13, Pretoria, South Africa.
- POLLARD, S.R., BIGGS, H. and DU TOIT, D.R. 2008. *Towards a Socio-Ecological Systems View of the Sand River Catchment, South Africa: An exploratory Resilience Analysis*. Water Research Commission Report: TT 364/08, Pretoria, South Africa.
- PORTER, M. E. 1985. *Competitive Advantage: Creating and Sustaining Superior Performance*. The Free Press, New York, US.
- POTSCHIN, M.B. and HAINES-YOUNG, R.H. 2011. Ecosystem services: exploring a geographical perspective. *Progress in Physical Geography*, 35(5):575-594.
- POWELL, M.J. 2009. *Restoration of degraded subtropical thickets in the Baviaanskloof Megareserve, South Africa: The role of carbon stocks and *Portulacaria afra* survivorship*. Unpublished MSc Thesis, Department of Environmental Science, Rhodes University, Grahamstown, South Africa.

- PROCHES, S and COWLING, R.M. 2007. Do insect distributions fit our biomes? *South African Journal of Science*, 103:258-261.
- RAIKES, P., JENSEN, S.P. and PONTE, S. 2000. *Global Commodity Chain Analysis and the French Filière Approach: Comparison and Critique*. CDR Working Paper, Working Paper Subseries on Globalisation and Economic Restructuring in Africa: ix. Centre for Development Research, Copenhagen, Denmark.
- RAMSAY, J. 2005. The real meaning of value in trading relationships. *International Journal of Operations and Production Management*, 25(6):549-565.
- RANDERS, J. 1980. *Elements of the System Dynamics Method*. MIT Press, Cambridge, UK.
- RAUDSEPP-HEARNE, C., PETERSON, G.D. and BENNETT, E.M. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *PNAS*, 107(11):5242-5247.
- RICHARDSON, G.P. 1997. Problems in causal loop diagrams revisited. *System Dynamics Review*, 13:247-252.
- RICHARDSON, G.P. 1996. *Modelling for Management*. Dartmouth, Aldershot, UK.
- RICHARDSON, G.P. 1994. Introduction: Systems thinkers, systems thinking. *System Dynamics Review*, 10(2/3):95-99.
- RICHARDSON, G.P. and Pugh, A.L. 1989. *Introduction to System Dynamics Modeling*. Pegasus Communications, Waltham, US.
- RICHARDSON, G. P. 1986. Problems with causal-loop diagrams. *System Dynamics Review*, 2(2):158-170.
- RICHMOND, B. 1994. Systems thinking/system dynamics: let's just get on with it. *System Dynamics Review*, 10:135–157.
- ROCKSTRÖM, J., STEFFEN, W., NOONE, K., PERSSON, Å., CHAPIN, F.S., LAMBIN, I.E., LENTON, T.M., SCHEFFER, M., FOLKE, C., SCHELLNHUBER, H., NYKVIST, B., DE WIT, C.A., HUGHES, T., VAN DER LEEUW, S., RODHE, H., SÖRLIN, S., SNYDER, P.K., COSTANZA, R., SVEDIN, U., FALKENMARK, M., KARLBERG, L., CORELL, R.W., FABRY, V.J., HANSEN, J., WALKER, B., LIVERMAN, D., RICHARDSON, K., CRUTZEN, P. and FOLEY, J. 2009. Planetary boundaries: exploring the safe operating space for humanity. *Ecology and Society*, 14(2):32.
- ROMERO, I. and TEJADA, P. 2011. A multi-level approach to the study of production chains in the tourism sector. *Tourism Management*, 32:297-306.

- RUST, I.C. and ILLENBERGER, W.K. 1989. *Geology and geomorphology of the Baviaanskloof*. In: Kerley, G.I.H. and Els, L.M. (eds). The Kouga-Baviaanskloof Complex: Conservation Status and Management Problems. Proceedings of a Workshop held at the University of Port Elizabeth. Chief Directorate: Nature and Environmental Conservation and University of Port Elizabeth. President Printers, Port Elizabeth, South Africa.
- RUTH, M. 1993. *Integrating economics, ecology and thermodynamics*. Springer, Dordrecht, Netherlands.
- RYAN, A.M. and SPASH, C.L. 2011. Is WTP an attitudinal measure? Empirical analysis of the psychological explanation for contingent values. *Economic Psychology*, 32:674-687.
- SAGOFF, M. 2011. The quantification and valuation of ecosystem services. *Ecological Economics*, 70:497-502.
- SALADIN, M., NEUBAUER, L., BOROWSKI-MAASER, I. and ROTTER, S. 2012. *ESAWADI – Ecosystem Services Approach for Water Framework Directive Implementation*. German Case Study Report. [Online]. Available: <http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.463.8785&rep=rep1&type=pdf>. [09/08/2015].
- SALLES, J.M. 2011. Valuing biodiversity and ecosystem services: Why put economic values on nature? *Comptes Rendus Biologies*, 334(5–6):469–482.
- SANBI (South African National Biodiversity Institute). 2007. *Strategic Water Source Areas (SWSA)*. Biodiversity GIS. [Online]. Available: <http://bgis.sanbi.org/NFEPA/SWSAmap.asp>. [07/10/2014].
- SARRIOT, E., MORROW, M., LANGSTON, A., WEISS, J., LANDEGGER, J. and TSUMA, L. 2015. A casual loop analysis of the sustainability of integrated community case management in Rwanda. *Social Science and Medicine*, 131:147-155.
- SCHAFFERNICHT, M. 2010. Causal loop diagrams between structure and behaviour: A critical analysis of the relationship between polarity, behaviour and events. *Systems Research and Behavioural Science*, 27:653-666.
- SCHMITZ, H. 2005. *Value chain analysis for policy makers and practitioners*. International Labour Office, Geneva, Switzerland.
- SCHOLL, G.J. 1995. Benchmarking the system dynamics community: research results. *System Dynamics Review*, 11:139–155.
- SEN, A.K. 1977. A critique of the behavioural foundations of economic theory. *Philosophy and Public Affairs*, 6(4):317-344.

- SENGE, P.M. 1990. *The Fifth Discipline: The Art and Practice of the Learning Organization*. Doubleday, New York, US.
- SENGE, P.M. and STERMAN, J.D. 1992. *Systems thinking and organizational learning: Acting locally and thinking globally in the organization of the future*. In: Kochan, T. and Useem, M. (eds). *Transforming organizations*. Oxford University Press, New York, US.
- SKOWNO, A. 2008. *Baviaanskloof Mega-Reserve Biodiversity Assessment: Version 3*. [Online]. Available: http://bgis.sanbi.org/baviaanskloof/BMR_Biodiversity_assessment.pdf. [05/05/2015].
- SOKAL, R.R. 1974. Classification: Purposes, Principles, Progress, Prospects. *Science*, 185:1115-1123.
- SOLOW, R.M. 1993. An almost practical step towards sustainability. *Resources Policy*, 16:162–72.
- STAHEL, A.W. 2005. Value from a complex dynamic system's perspective. *Ecological Economics*, 54:370-381.
- STAMM, A. and VON DRACHENFELS, C. 2011. *Value chain development: approaches and activities by seven UN agencies and opportunities for interagency cooperation*. International Labour Office, Geneva, Switzerland.
- STAMM, A. 2004. *Value chains for Development Policy: Challenges for Trade Policy and the Promotion of Economic Development*. German Development Institute, Eschborn, Germany.
- STAVE, K.A. 2003. A system dynamics model to facilitate public understanding of water management options in Las Vegas, Nevada. *Environmental Management*, 67:303-313.
- STERMAN, J. 2000. *Business Dynamics: Systems Thinking and Modeling for a Complex World*. McGraw-Hill, Boston, US.
- STEWART, T. J., JOUBERT, A. R., SCOTT, L. and LOW, T. 1997. *Multiple criteria decision analysis: Procedures for consensus seeking in natural resources management*. Water Research Commission Report: 512/1/97, Pretoria, South Africa.
- STURGEON, T.J. 2001. How do we define value chains and production networks? *IDS Bulletin*, 32(3):9–18.
- TALLEC, F. and BOCKEL, L. 2005. *Commodity Chain Analysis: Constructing the Commodity Chain Functional Analysis and Flow Charts*. EASYPol Module: 043, Food and Agriculture Organization of the United Nations, Rome.

- TALLIS, H. and POLASKY, S. 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. *Annals of the New York Academy of Sciences*, 1162:265-283.
- TEEB 2010. The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. Edited by Kumar, P. Earthscan, London and Washington, UK and US.
- TIETENBERG, T. and LEWIS, L. 2010. *Environmental Economics and Policy* (6e). Pearson, Cape Town, South Africa.
- TUFTE, E.R. 1990. *Envisioning Information*. Graphics Press, Cheshire, UK.
- TURNER, R.K., PAAVOLA, J., COOPER P., FARBER, S., JESSAMY, V. and GEORGIU, S. 2003. Valuing nature: Lessons learned and future research directions. *Ecological Economics*, 46:493-510.
- TURNER, R.K., PEARCE, D. and BATEMAN, I. 1994. *Environmental economics: An elementary introduction*. Harvester Wheatsheaf, New York, US.
- TURPIE, J., LANNAS, K., SCOVRONICK, N. and LOUW, A. 2010. *Wetland Valuation Volume I. Wetland ecosystem services and their valuation: A Review of current understanding and practice*. Water Research Commission Report: TT 440/09, Pretoria, South Africa.
- USAID (The United States Agency for International Development). 2007. *Value chain framework*. [Online]. Available: <http://microlinks.kdid.org/goodpractice-center/value-chain-wiki/value-chain-framework>. [11/03/2015].
- USNRC (United States National Research Council). 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision Making*. The National Academies Press, Washington, US.
- VAN DEN BELT, M. 2000. *Mediated modelling: a system dynamics approach to environmental consensus building*. Island Press, Washington, US.
- VAN DEN BERG, J., INGRAM V.J., BOGAARDT, M.J. and HARMS, B. 2013. *Integrating ecosystem services into the tropical timber value chain: Dutch policy options from an innovation system approach*. The Statutory Research Tasks Unit for Nature & the Environment WOt-werkdocument: 344, Wageningen, Netherlands.
- VAN DER BURG, L. 2008. *Valuing the benefits of restoring the water regulation services, in the subtropical thicket biome: a case study in the Baviaanskloof-Gamtoos watershed, South-Africa*. Unpublished MSc Thesis, Wageningen University, Wageningen, Netherlands.

- VAN JAARSVELD, A.S., BIGGS, R., SCHOLE, R.J., BOHENSKY, E., REYERS, B., LYNAM, T., MUSVOTO, C. and FABRICIUS, C. 2005. Measuring conditions and trends in ecosystem services at multiple scales: the Southern African Millennium Ecosystem Assessment (SAfMA) experience. *Philosophical Transactions of the Royal Society*, 360:425-441.
- VAN LUIJK, G., COWLING, R.M., RIKSEN, M.J.P.M. and GLENDAY, J. 2013. Hydrological implications of desertification: degradation of South African semi-arid subtropical thicket. *Journal of Arid Environments*, 91:14-21.
- VAN WILGEN, B.W., REYERS, B., LE MAITRE, D.C., RICHARDSON, D.M. and SCHONEGEVEL, L. 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Environmental Management*, 89:336-349.
- VENNIX, J.A.M., ANDERSEN, D.F. and RICHARDSON, G.P. 1997. Group model building, art and science. *System Dynamics Review*, 12(2):103-106.
- VENNIX, J.A.M. 1996. *Group model building: Facilitating team learning using system dynamics*. Wiley, New York, US.
- VIGERSTOL, K.L. and AUKEMA, J.E. 2011. A comparison of tools for modeling freshwater ecosystem services. *Journal of Environmental Management*, 92:2403–2409.
- VIHERVAARA, P., RÖNKÄ, M. and WALLS, M. 2010. Trends in Ecosystem Service Research: Early Steps and Current Drivers. *Ambio*, 39:314–324
- VILLA, F., BAGSTAD, K. J., VOIGHT, B., JOHNSON, G. W., PORTELA, R. HONZAK, M. and BATKER, D. 2014. A methodology for adaptable and robust ecosystem service assessment. *PLoS ONE*, 9(3):1-18.
- VILLA, F., BAGSTAD, K., JOHNSON, G. and VOIGT, B. 2011. Scientific instruments for climate change adaptation: estimating and optimizing the efficiency of ecosystem services provision. *Economía Agraria y Recursos Naturales*, 11(1):54–71.
- VOSS, C., TSIKRIKTSIS, N. and FROHLICH, M. 2002. Case research in operations management. *International Journal of Operations & Production Management*, 22(2):195-219.
- WALKER, B. and SALT, D. 2006. *Resilience thinking: sustaining ecosystems and people in a changing world*. Island Press, Washington, US.
- WALKER, B., CARPENTER, S., ANDERIES, J., ABEL, N., CUMMING, G., JANSSEN, M., LEBEL, L., NORBERG, J., PETERSON, G.D. and PRITCHARD, R. 2002. Resilience management in social-ecological systems: a working hypothesis for a participatory approach. *Conservation Ecology*, 6(1):14.

- WALLACE, K.J. 2008. Classification of ecosystem services: Problems and solutions. *Biological Conservation*, 139:235–246.
- WEISS, G., PETTENELLA, D., OLLONQVIST, P. and SLEE, B. 2011. *Innovation in Forestry Territorial and Value Chain Relationships*. CABI, Wallingford, UK.
- WELMAN, P.I. AND BARNARD, H.C. 2007. *Zaaimanshoek Bulk Water Supply*. Groundwater Consulting Services Hydrogeological Investigation Draft Report, Johannesburg, South Africa.
- WILSON, M.A. and HOEHN, J.P. 2006. Valuing environmental goods and services using benefit transfer: the state-of-the art and science. *Ecological Economics*, 60:335-342.
- WOLSTENHOLME, E.F. 1999. Qualitative vs quantitative modelling: the evolving balance. *Journal of the Operational Research Society*, 50:422-428.
- WOOD, A. 2001. Value chains: An economist's perspective. *IDS Bulletin*, 32(3):1-5.
- WORKING FOR WETLANDS PROGRAMME. 2015. *Rehabilitation Plan for the Baviaanskloof Wetland Project, Eastern Cape: Planning Year 2015/2016*. Working for Wetlands Rehabilitation Programme Report: 109664/10065, Port Elizabeth, South Africa.
- WRI (World Resources Institute). 2000. *World resources 2000-2001: people and ecosystems, the fraying web of life*. United Nations Development Programme and World Bank. World Resources Institute, Washington, US.
- WRG (The 2030 Water Resources Group). 2009. *Charting our water future: economic frameworks to inform decision-making*. [Online]. Available: <https://www.2030wrg.org/team/charting-our-water-future/>. [13/12/2015].
- WUNDER, S. 2007. The efficiency of payments for environmental services in tropical conservation. *Conservation Biology*, 21(1):48-58.
- WUNDER, S. 2005. *Payments for environmental services: some nuts and bolts*. CIFOR Occasional paper: 42. Center for International Forestry Research, Bogor, Indonesia.
- YIN, R.K. 2012. *Applications of Case Study Research*. SAGE Publications, Los Angeles, US.
- YIN, R.K. 1994. *Case study research design and methods* (2e). SAGE Publications, Thousand Oaks, US.
- YONG, W.U. 2005. *Groundwater recharge estimation in Table Mountain Group aquifer systems with a case study of Kammanassie area*. Unpublished PhD thesis, Department of Earth Sciences, University of the Western Cape, Cape Town, South Africa.

ZHANG, B. and GOVINDARAJU, R.S. 2000. Prediction of watershed runoff using Bayesian concepts and modular neural networks. *Water Resources Research*, 36(3):753-762.

APPENDIX



Photo: J.M. Rawlins

1. Causal Loop Diagram Development Contributing Participants

Table 1: Expert workshop participants.

Prof. Tally Palmer	Professor and director of Unilever Centre for Environmental Water Quality, Rhodes University.
Prof. Fred Ellery	Professor and head of the Department of Geography, Rhodes University.
Dr. Tony Palmer	Specialist scientist and Research Associate, Agricultural Research Council, Animal Production Institute.
Rebecca Powell	PhD candidate in the Department of Environmental Science, Rhodes University.
Julia Glenday	PhD candidate at the University of Santa Barbara and LivingLands research assistant.
Dr. Willem De Lange (Supervisor)	Senior researcher in the Natural Resources and Environment unit, CSIR.
Prof. Gavin Fraser (Supervisor)	Professor and head of the Department of Economics and Economic History, Rhodes University.
Jonathan Rawlins (MSc Candidate)	MSc candidate in the Department of Economics and Economic History, Rhodes University.

Table 2: Professional verification meeting participants.

Prof. Tally Palmer	Professor and director of Unilever Centre for Environmental Water Quality, Rhodes University.
Prof. Jen Snowball	Professor in the Department of Economics and Economic History, Rhodes University.
Dr. Jane Turpie	Senior Lecturer for Percy FitzPatrick Institute at the University of Cape Town and Director of Anchor Environmental Consultants.
Dr. Patrick O'Farrell	Senior researcher at the Council for Scientific and Industrial Research, Stellenbosch.
Dr. David Le Maitre	Senior researcher at the Council for Scientific and Industrial Research, Stellenbosch.
Dieter Van Den Broeck	Co-founder of Living Lands and landscape orchestrator at Commonland.
Julia Glenday	PhD candidate at the University of Santa Barbara and LivingLands research assistant.
Stephie Mendlesohn	Knowledge broker and research coordinator at Living Lands, Baviaanskloof.
Pieter Kruger	Owner and manager of Zandvlakte Farm, Baviaanskloof.

2. Aquatic Ecosystem Service Descriptions and Classifications

Table 3: Final aquatic ecosystem services descriptions, classifications and beneficiary groups.

Final Aquatic Ecosystem Service	Description	CICES Classification	Beneficiary Group
Water Provision	The volume of water provided by the ecosystem that is suitable for consumptive use. This is determined by the <u>yield</u> and <u>quality</u> of the water.	Section: Provisioning Division: Nutrition Group: Water Class: Surface/Ground water for drinking Class Type: <i>By amount, type</i>	Catchment and downstream domestic water users.
Aquatic Ecosystem Health	Overall condition of the ecosystem. The sum of all biotic and abiotic components and their systematic interactions that contribute towards ecosystem function.	n/a	Conservationists and tourists.
Flood Attenuation	The ability of an ecosystem to retain water in situ and release it over time, reducing flood occurrences and damage.	Section: Regulation and maintenance Division: Mediation of flows Group: Liquid flows Class: Flood protection Class Type: <i>By reduction in risk, area protected</i>	Catchment and downstream residents.

Table 4: Intermediate aquatic ecosystem service descriptions, units of measurement and causal linkages.

Intermediate Aquatic Ecosystem Service	Description	General Unit of Measurement	Outgoing Causal Links
Groundwater Flow	Water present beneath the Earth's surface over time.	Volume/Time	Water Yield (+)
Surfacewater Flow	Water present on the Earth's surface over time, this includes channel and overland flow.	Volume/Time	Water Yield (+) Flow Velocity (+)
Water Yield	Average amount of freshwater that runs off in the catchment, includes all ground and surface water.	Volume	Water Provision (+) Aquatic Ecosystem Health (+)
Water Quality	The state of multiple chemical and physical variables that individually and collectively determine biotic responses and potential water uses.	Multiple Units (e.g. micrograms of nitrates/Volume)	Water Provision (+) Aquatic Ecosystem Health (+)
Evaporation Rate	Process of water changing state from liquid to water vapour due to increased temperature and/or decreased pressure.	Volume/Time	Groundwater Flow (-) Surfacewater Flow (-) Soil Salinisation (+)

Transpiration Rate	Process of plants converting water into water vapour which is released into the atmosphere.	Volume/Time	Groundwater Flow (-) Soil Salinisation (+) Crop Water Demand (+)
Infiltration Rate	Process of water on the ground surface entering the soil.	Volume/Time	Groundwater Flow (+) Surfacewater Flow (-)
Crop Water Demand	Amount of water required for a cultivated crop to meet the water loss through evapotranspiration.	Volume	Irrigation (+)
Atmospheric Temperature	Measure of the temperature at the surface of the Earth.	Degrees Celsius	Evaporation Rate (+) Transpiration Rate (+)
Interception Rate	The process of leaves and branches of plants preventing precipitation from reaching the ground (soil).	Volume/Time	Groundwater Flow (-) Surfacewater Flow (-)
Roughness	Floodplain/channel shape and material texture that affects flow velocity through friction.	Manning's Roughness Coefficient (Limerinos, 1970; Li and Zhang, 2001)	Floodplain Capacity (+) Flow Velocity (-)
Floodplain Capacity	Space available in the floodplain and surrounding area capable of retaining water.	Volume	Flood Attenuation (+)
Flow Velocity	The rate at which water flows in a channel or over the surface of the Earth.	Velocity	Erosion (+) Flood Attenuation (-)

Natural Vegetation	Amount of natural vegetation (biomass) present.	Mass	Water Quality Amelioration (+) Transpiration Rate (+) Interception Rate (+) Infiltration Rate (+) Roughness (+) Livestock Units (+) Soil Stability (+)
Grazing	Amount of natural vegetation consumed by livestock units over time (grazing and/or browsing).	Volume/Time	Natural Vegetation (-)
Soil Salinisation	Biophysical process of increasing soil salt content.	Mass/Volume	Salinity (+)
Water Quality Amelioration	Variety of biophysical processes that improve overall water quality. These include but are not limited to the dilution, assimilation and transport of waste water and pollution as well as sediment and nutrient retention.	Multiple Units (e.g. Volume of sediment retained/Time)	Sediment Load (-)) Pesticide Content (-) Nutrient Load (-) Microbial Content (-) Salinity (-)

Rainfall	The quantity of rain falling within a given area over a specific time period.	Volume	Natural Vegetation (+) Irrigation (-) Interception Rate (+) Surfacewater Flow (+)
Erosion	Action of exogenic processes which remove soil and rock from the Earth's surface.	Mass/Time	Sediment Load (+) Floodplain Capacity (-) Aquatic Ecosystem Health (-)
Soil Stability	Ability of the soil to resist disintegration when disruptive forces associated with different types of erosion are applied (Kemper and Rosenau, 1986).	Normalised Stability Index (Six <i>et al.</i> , 2000).	Erosion (-)
Salinity	Measure of all the salts and substances dissolved in water.	Mass/Volume	Water Quality (-)
Microbial Content	Pathogenic microorganisms in surface and ground water.	Mass/Volume	Water Quality (-)
Nutrient Load	Phosphates and nitrates in surface and ground water.	Mass/Volume	Water Quality (-)
Pesticide Content	Pesticides and herbicides in surface and ground water.	Mass/Volume	Water Quality (-)
Sediment Load	Solid matter carried in suspension in surface and ground water.	Mass/Volume	Water Quality (-)

Table 5: Anthropogenic variable descriptions, units of measurement and causal linkages.

Anthropogenic Variables	Description	General Unit of Measurement	Outgoing Causal Links
Livestock Units	Number of livestock units in a specific area.	Livestock Units/Area	Crops (+) Groundwater Pumping (+) Grazing (+) Aquatic Ecosystem Health (-)
Groundwater Pumping	Artificial transport of groundwater to the surface for irrigation or livestock use over time.	Volume/Time	Groundwater Flow (-)
Crops	Cultivated plants grown for commercial or subsistence purposes (fodder and/or cash crops).	Area	Fertiliser/Pesticide Application (+) Transpiration Rate (+) Irrigation (+) Natural Vegetation (-)
Irrigation	Artificial application of water to land or soil.	Volume/Time	Groundwater Pumping (+) Interception Rate (+) Abstraction (+)
Abstraction	Process of extracting surface water for anthropogenic use.	Volume/Time	Surfacewater Flow (-)
Fertiliser/Pesticide Application	Fertiliser and/or pesticide applied to cultivated crops over time.	Volume/Time	Pesticide Content (+) Nutrient Load (+)

3. Examples of Causal Loop Diagram Development Versions

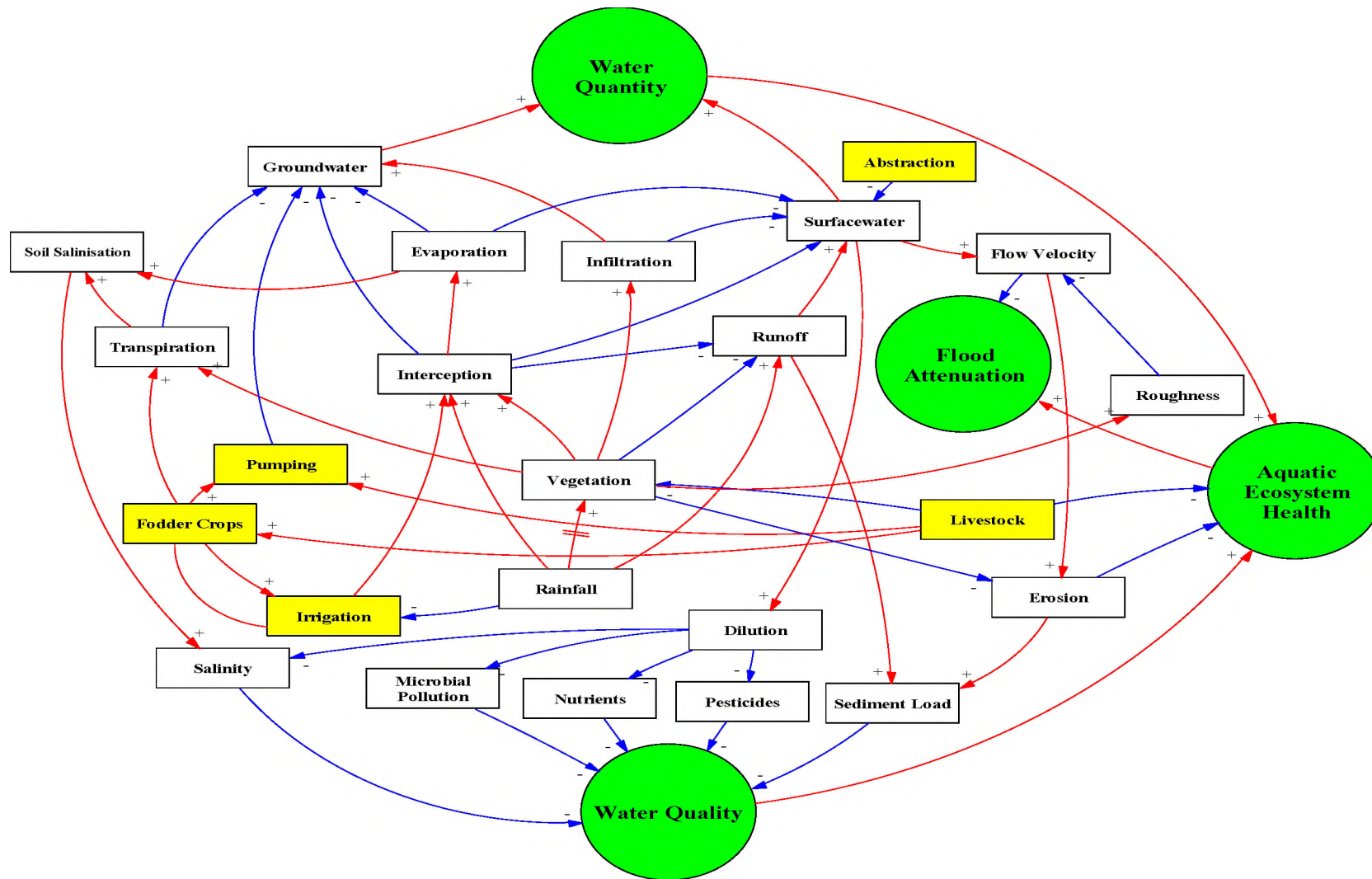


Figure 1: First aquatic ecosystem service causal loop diagram version.

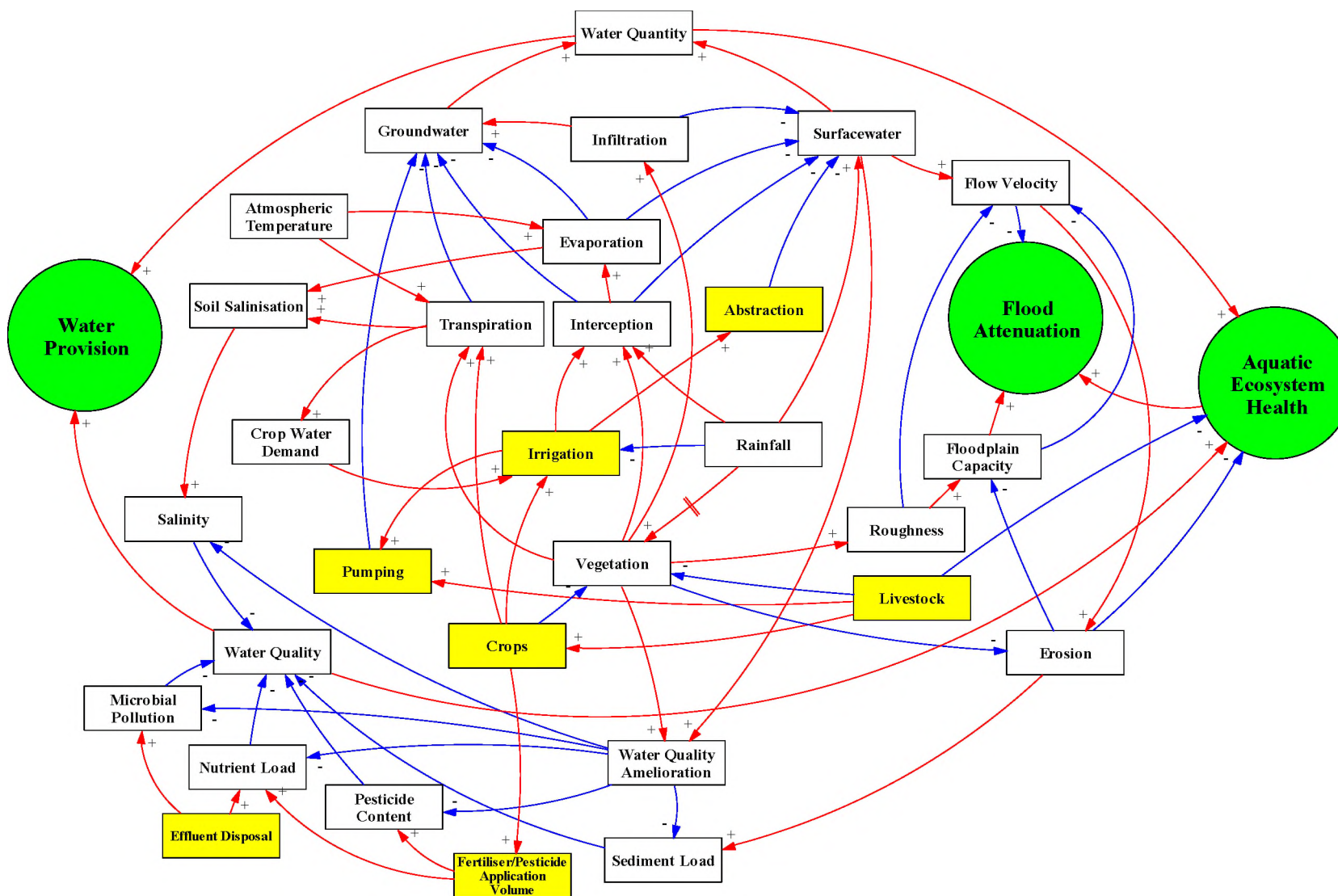


Figure 2: Sixth aquatic ecosystem service causal loop diagram version.

