ASSESSING AQUATIC ECOSYSTEM SERVICES VALUE CHAINS AND MARKETS IN SOUTH AFRICA: SOME CASE STUDIES

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by

W de Lange
(CSIR)

J Rawlins & G Fraser
(Rhodes University)

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Water Research Commission
Private Bag X03
Gezina 0031

orders@wrc.org.za or download from www.wrc.org.za

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EXECUTIVE SUMMARY

We still have a limited understanding of the value chains, markets and the actual economic value of ecosystem services from aquatic ecosystems. Different studies have developed various approaches for determining the economic value of these benefits, and of the associated natural capital. Most confirm that the value of aquatic ecosystems lies in the sustained nett benefits derived from the many ecosystem services they supply; including various ecological functions, products for direct and indirect human consumption, energy, aesthetic and recreational benefits, and assimilative capacity of the residues of human activities. However, the geographic, cultural and economic differences between countries or nations have resulted in different views that affect the market potential of ecosystem services from aquatic ecosystems.

This study focused on identifying key ecosystem services and their forward linkages, understanding how to improve market access to such services, and creating or improving the value chains in the South African context. The research is intended to help identify opportunities for improvements that benefit society more broadly. It is anticipated that the study will be useful to land use planners, designers of infrastructure and town planners.

The study explored aquatic ecosystem services (AESs) in South Africa within the context of a value chain assessment aimed at the following:

- Investigating the forward linkages in the value chains of AESs and their markets in South Africa.
- Identifying challenges and opportunities in the value chains of existing markets.
- Investigation the ripple effects of AESs in South Africa.
- Recommending ways to improve the value chains of AESs.

The project explored AESs in South Africa by means of a desktop study. All known AESs in South Africa were systematically accounted for and their current value chains assessed to provide practical recommendations for developing new markets or strengthening existing markets for these goods and services (depending on the chain).

A causal loop diagram (CLD) formed the basis for presenting and analysing the value chains of AES and graphically presenting the supply and user/consumer sides (including beneficiaries) of the value chain, along with the route the goods and/or services follow. The work was done in two case studies. The CLD made it easier to identify the transactional route for each value chain, which in turn made it possible to identify potential inefficiencies in the chain. The consequent assessment of these inefficiencies informed the basis for recommending improvements to a market-making process for AESs.

The report presents relevant literature reviews, the methodological approach followed to construct the CLD for AESs in a study area, as well as the CLD itself. A scenario analysis presents the ripple effects of external shocks on the CLD and associated recommendations to improve (increase) the resilience of AESs in the study area. This information is used to argue the case for taking AESs to market, which is done by a market-based approach explained in terms of a market-making process for a selected AES.

Taking AES to market as a way of incentivising increased private sector investment to improve the service seems possible in theory, and could be done in several ways. This study focused on the market-making process of water pollution permits. Although a tradable permit system is, due to its nature, a complex instrument in terms of implementation, it has immense potential to effectively mitigate pollution once up and running. This is mainly because polluters differ in their ability to abate their pollution – some can do it easily and cheaply, while others would find it to be more difficult and costly. The freedom to trade pollution ‘entitlements’ gives an incentive to polluters to consider abatement (since they can

sell their surplus quotas) while others face the cost of having to purchase permits. For society, the existence of tradable permits enables pollution abatement to be achieved in a cost-effective way. Over time, pollution standards can be tightened, thereby increasing the value of the permits and the pressure on market participants to pollute less.

This report presents fundamental conditions and requirements for a water pollution permit system along with the processes that will facilitate implementation for South African river systems and their associated AESs. Due to the novelty of the work, the focus was on the preliminary design of the system – a process which is ongoing. Consequently, no information was put forward regarding the political process that is required to facilitate acceptance among those affected. However, it should be clear that in order to have a functional market for water pollution permits, enough market participants (i.e. polluters) should exist (i.e. enough pollution should be present), which means that there should be many pollution sources (polluters) affecting the same parameter (e.g. nitrogen, phosphorous, biochemical oxygen demand, salinity) within the same catchment, while having significant differences in abatement costs so that beneficial trade becomes possible. Furthermore, establishing a legal entity (scheme administrator) responsible for monitoring and data capturing, enforcement mechanisms, information for conflict resolution, trade facilitation and system evaluation, along with a solid scientific understanding of the pollution factors system responses, will provide a solid basis for trade. However, it is imperative that the scheme administrator keeps administrative requirements, approval procedures and requirements to trade to a minimum since high transaction costs reduces willingness to trade.

It is also advisable that the existing functioning system of water pollution control should be in place to design and implement a market for tradable pollution permits. Such a system not only provides the basic context to design the permit system (ensuring compatibility), but it also streamlines the proposed system with existing data sources. It should be noted that the pollution permit system is not a failsafe way to decrease the absolute level of pollution within a river, but should form part of an integrated strategy and in conjunction with command-and-control regulations, to mitigate pollution problems. Here, the monetary value of the impact of the pollution not only provides a "management budget" for pollution mitigation strategies, but also enables the calculation of a reserve price for pollution permits. Permits have a particular role to play to avoid eutrophic conditions via relocating the pollution problem to sub-areas within the river that can best cope with the load at a particular time period.

It is evident from the findings that CLDs have the potential to facilitate an 'alternative' value chain analysis where traditional approaches to value chain analyses are unsuitable. 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 ecosystem service value chain analysis (ESVCA) framework as developed in this study 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. It also provides a 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, thereby allowing for a proactive approach towards ecosystem management that is geared towards improving the provision of chosen AESs while increasing the system understanding for relevant stakeholders and decision makers. Private and public entities that rely on the provision of ecosystem services have the potential to take advantage of this approach to recognise potential opportunities and threats within the value chains of these ecosystem services. Identifying the most efficient methods of improving ecosystem service 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 that 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.

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 socioecological systems while simultaneously assessing assumptions of policy and practice aimed at improving human well-being through improved provision of essential ecosystem services. This is done by using the ecosystem service, CLD and value chain concepts to provide information to support long-term sustainable management of the socioecological systems upon which all life depends.

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 illustrated 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 was used to illustrate ways to improve AES provision and associated value chains effectively addressing all four research aims of the study.

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INVESTIGATION OF AQUATIC ECOSYSTEM SERVICES, THEIR VALUE CHAIN, AND MARKETS IN SOUTH AFRICA

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Mr Bonani Madikizela	WRC
Dr M Stanley Liphadzi	WRC
Dr Chris Moseki	WRC
Mr Kyle Harris	Prime Africa
Prof Geoff Antrobus	Rhodes University
Ms Tracy Cumming	SANBI
Mr John Dini	SANBI
Dr Pauline Chivenge	UKZN
Dr Mao Amis	Africa Green
Dr Jackie Crafford	Prime Africa

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GLOSSARY OF KEY TERMS

Catalytic finance(ing) – finance aimed at stimulating/facilitating additional investment in goods or services. Within the context of this project it could be considered as ‘seed’ funding to enable projects to seek further investment from private and public sources.

Causal loop diagram – a qualitative diagramming language aimed at graphically illustrating feedback-driven systems.

Command and control – enforced legislation.

Compliance market – where public regulation requires payment for the use of ecosystem services (e.g. mandatory carbon emission trading for certain industries).

Ecological infrastructure – naturally functioning ecosystems that deliver valuable goods and services to people.

Ecosystem service – the flow of benefits derived from ecological infrastructure.

Green economy – a multi-faceted concept emphasising sustainability principles for economic development pathways.

Government-mediated market – where the government is the intermediate party that collects payments from users and distributes the funds to the service providers (e.g. payments for water services).

Market – a social construct facilitating utility exchange that is overseen by an accepted protocol.

Non-excludability – when it is impossible to create property rights (or when the cost of enforcement of property rights are too high) to exclude someone from benefitting from a good or service.

Non-rivalry – when the use of a good or service by an individual does not affect the quality or quantity of the same good or service to others.

Payments for ecosystem services (PES) – a concept referring to the need for compensating those investing in the maintenance and restoration of ecological infrastructure by those who benefit from such investment.

PES scheme – a specific protocol facilitating compensation of those investing in the maintenance and restoration of specific ecological infrastructure by those who benefit from such investment.

Public goods – goods that present characteristics of non-rivalry in consumption, and non-excludability in provision.

Voluntary market – where companies voluntarily decide to compensate their impact on the environment by purchasing compensatory credits (e.g. voluntary carbon emission credits) since such investment will either: provide a business opportunity; realise cost savings; secure operations (licensing for mines); appreciates the value of the company’s assets or decreases the company’s risk profile.

LIST OF ABBREVIATIONS

AES	Aquatic Ecosystem Services
ANT	Actor-network Theory
BMR	Baviaanskloof Mega Reserve
CAMPFIRE	Communal Areas Management Programme for Indigenous Resources
CICES	Common International Classification of Ecosystem Services
CLD	Causal Loop Diagram
CSI	Corporate Social Investment
CVM	Contingent Valuation Method
DAWASCO	Dar es Salaam Water Supply and Sewerage Corporate
DWAF	Department of Water Affairs and Forestry
EPWS	Equitable Payments for Watershed Services
ESVCA	Ecosystem Service Value Chain Analysis
GEF	Global Environment Facility
KL	Kwanza Limited
masl	Metres Above Sea Level
MA	Millennium Ecosystem Assessment
MMODS	A Mental Model of a Dynamic System
NGO	Non-governmental Organisation
NVA	Network Value Analysis
PES	Payment for Ecosystem Services
SEEA	System of Environmental Economic Accounting
SFD	Stock/Flow Diagram
SLM	Sustainable Land Management
SWSA	Strategic Water Source Area
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TMG	Table Mountain Group
TNC	The Nature Conservancy
US	United States
USA	United States of America
VCA	Value Chain Analysis
WCS	Wildlife Conservation Society
WfW	Working for Water
WRC	Water Research Commission
WTA	Willingness to Accept
WTP	Willingness to Pay

1 INTRODUCTION

We still have a limited understanding of the value chains, markets and the actual economic value of ecosystem services from aquatic ecosystems. Different studies have developed various approaches for determining the economic value of these benefits, and of the associated natural capital. Most confirm that the value of aquatic ecosystems lies in the sustained nett benefits derived from the many ecosystem services they supply; including various ecological functions, products for direct and indirect human consumption, energy, aesthetic and recreational benefits, and assimilative capacity of the residues of human activities. However, the geographic, cultural and economic differences between countries or nations have resulted in different views that affect the market potential of ecosystem services from aquatic ecosystems.

This study focused on identifying key ecosystem services and their forward linkages, understanding how to improve market access to such services, and creating or improving the value chains in the South African context. The research is intended to help identify the opportunities for improvements that benefit society more broadly. It is anticipated that the study will be useful to land use planners, designers of infrastructure and town planners.

Specific aims of the study included:

- Investigating the forward linkages in the value chains of aquatic ecosystem services (AESs) and their markets in South Africa.
- Identifying challenges and opportunities in the value chains of existing markets.
- Investigation the ripple effects of AESs in South Africa.
- Recommending ways to improve the value chains of AESs.

A causal loop diagram (CLD) formed the basis for presenting and analysing the value chains of AES and graphically presenting the supply and user/consumer sides (including beneficiaries) of the value chain, along with the route the goods and/or services follow. The work was done in two case studies. The CLD enabled easier identification of the transactional route for each value chain, which in turn enabled the identification of potential inefficiencies in the chain. The consequent assessment of these inefficiencies informed the basis for recommending improvements to a market-making process for AESs. The report presents relevant literature reviews, the methodological approach being followed to construct the CLD for AESs in a study area, as well as the CLD itself. A scenario analysis presents the ripple effects of external shocks on the CLD and associated recommendations to improve (increase) the resilience of AESs in the study area. This information is used to argue the case for taking AESs to market, which is done by a market-based approach explained in terms of a market-making process for a selected AES.

The report presents relevant literature reviews (Sections 2, 3 and 4), the methodological approach followed to construct the CLD for AESs in the study area (section 5), the study area (Section 6) and the CLD (section 7). The scenario analysis (Section 8) presents the ripple effects of external shocks (Section 9) on the CLD and associated recommendations to improve (increase) the resilience of AESs in the study area (Section 10). This information is used to argue the case for taking AESs to market (Section 11), which is done by a market-based approach (Section 12) explained in terms of a market-making process (Section 13) for a selected AES. The report concludes with a short discussion and some key recommendations (Section 14).

2 AQUATIC ECOSYSTEM SERVICES

2.1 The Concept of Ecosystem Services and Definitional Issues

The theory of ecosystem services has developed into an important model linking human welfare with the functioning of natural ecosystems (Fisher et al., 2009; TEEB, 2010). Relevant and accurate policy and management decision-making relies largely on the understanding of these linkages (Costanza et al., 1997; MA, 2005; TEEB, 2010). Despite numerous attempts to develop a standardised classification scheme for ecosystem services, there still is no consistent, agreed-upon definition for the term (Boyd & Banzhaf, 2007; Fisher et al., 2009). Research focusing on ecosystem services has increased dramatically recently resulting in a myriad of proposed ecosystem service definitions for various applications (De Groot et al., 2010b; Haines-Young & Potschin, 2014; Vihervaara et al., 2010). Initially, Westman (1977) described how the benefits in terms of social value provided by ecosystems have the potential to be enumerated so more informed management and policy decisions can be made. These social benefits were termed ‘nature’s services’. It was not until Ehrlich and Ehrlich (1981) that the formal term for ecosystem services was coined. Mooney and Ehrlich (1997) further describe the history of the term and its transdisciplinary conceptualisation. However, there is little information available in the literature distinguishing how ecosystem services should be defined (Barbier, 2007; Boyd, 2007; Fisher et al., 2009; Wallace, 2008).

Some of the most widely used and cited definitions include:

- “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” (SEEA, 2012: 164).
- “The conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” (Daily, 1997: 3).
- “The benefits human populations derive, directly or indirectly, from ecosystem functions” (Costanza et al., 1997: 253).

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 is 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).

The general concept of ecosystem services requires defining an ‘ecosystem’, and distinguishing between these contested terms with specific reference to the application at hand (Carpenter et al., 2009; Costanza et al., 2014; COWI, 2014; Fisher et al., 2009; Nunes et al., 2014; ONEMA, 2011; TEEB, 2010; Wallace, 2008). The CBD (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”, which is the commonly accepted definition. The above-mentioned SEEA (2012) definition of ‘ecosystem services’ has gained increased support in recent times due to being aware of the need for such distinctions in terminology (COWI, 2014; Fisher et al., 2009).

ONEMA (2011) makes a minor distinction between ‘ecosystem services’ and ‘ecological services’. ‘Ecological services’ is preferred as it can be applied to various composite spatial units that 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 affected by processes that function over variable time scales (SEEA, 2012).

Boyd and Banzhaf (2007: 619) provide an alternative definition of final ecosystem services 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 streamline alternative to the aforementioned broad definitions. This definition highlights how ecosystem services are not the ‘benefits’ humans receive from ecosystems, but rather the ecological components consumed or enjoyed directly to yield human well-being (Boyd & 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 and elucidates that the concepts of ‘services’ and ‘benefits’ are not identical (Fisher et al., 2009).

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

Johnston and Russell (2011), and Landers and Nahlik (2013) note how most ecosystem service analyses fail to distinguish effectively between final ecosystem services and ecological functions, and processes that provide benefits indirectly, which are referred to as ‘intermediate ecosystem services’. Two primary sources of ambiguity arise when classifying values associated with final vs intermediate ecosystem services. Firstly, the lack of widely applicable, simple guidelines to aid in deriving replicable and consistent ecosystem service classification and, secondly, the application of universal final ecosystem service typologies that apply to all beneficiaries (Johnston & Russell, 2011).

The concept of humans as beneficiaries to these ‘environmental services’ means that these natural ‘functions’ and/or ‘processes’ are considered to be ecosystem services (Fisher et al., 2009; Limburg et al., 2002; ONEMA, 2011). However, ecosystem ‘structure’, ‘function’, and ‘services’ are not identical. Most ecosystem ‘structures’ and ‘processes’ do provide ecosystem services, but they are not necessarily the same thing. For example, Fisher et al. (2009), Limburg (2009), and Johnston and Russell (2011) describe how nutrient cycling is an example of a service that is used indirectly by people as one of the outcomes is clean water, which is a service that people use directly. Thus, clean drinking water can be considered to be a final ecosystem service or direct ‘benefit’ derived from different intermediate ecosystem services. Hence, ecosystem service science is moving towards the notion of differentiated intermediate and final ecosystem services, which is important to avoid double-counting of services among other ambiguities (COWI, 2014; Johnston & Russell, 2011; ONEMA, 2011; SEEA, 2012).

Figure 1 illustrates the interactions required to produce human well-being, which is an integral factor when defining ecosystem services. 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 subsystem of the environment because the natural environment provides all inputs into the economy, and all by-products and waste return the environment eventually (O’Neill et al., 2010). Figure 1 demonstrates how ecosystem services are the relative contribution of natural capital towards human well-being, despite not flowing directly. Thus, when addressing ecosystem services, it is imperative to use a transdisciplinary approach as there are a multitude of factors that determine how ecosystem services contribute towards human well-being (Carpenter et al., 2009; Costanza et al., 2014; COWI, 2014; MA, 2005; Van Wilgen et al., 2008; Wallace, 2008).

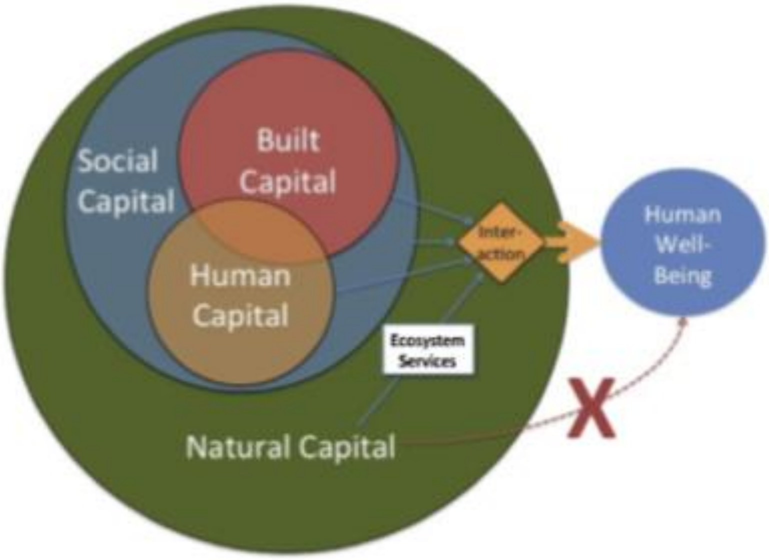

Figure 1: Interaction between built, social, human and natural capital required to produce human well-being (Costanza et al., 2014)

Many sources consider biodiversity to be the most important component of ecosystem services because it is a property of ecosystems as well as an ecosystem output, which is valued by humans as an ecosystem service in itself (De Groot et al., 2002; Van den Berg et al., 2013). Veeneklaas (2012) emphasises the complexity of the relationship between biodiversity and ecosystem services. High levels of biodiversity do not necessarily produce useful goods or services that contribute towards human well-being and vice versa (Van den Berg et al., 2013). Similarly, maintaining ecosystem services does not necessarily conserve biodiversity (Veeneklaas, 2012; TEEB, 2010). Biodiversity is the composite component that underpins the four categories of ecosystem services in various ways (TEEB, 2010).

The ecosystem service cascade (Figure 2) further illustrates the differentiation between ecosystem 'structure', 'function', 'service', 'benefit' and 'value'. The differentiation between these concepts is linked to the 'production boundary' that graphically illustrates a simplification of the interface between social and economic systems and the natural environment (COWI, 2014; SEEA, 2012). The ecosystem service cascade is a transdisciplinary and iterative analytical process used to systematically summarise much of the logic that underpins the modern ecosystem service paradigm (COWI, 2014; Potschin & Haines-Young, 2011).

The purpose of the ecosystem service cascade is to highlight the necessary elements that need to be considered in any ecosystem service analysis as well as the relationships between them (Potschin & Haines-Young, 2011). This model has been discussed and adapted by others (e.g. De Groot et al., 2010 (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 & Haines-Young, 2011). '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 & Haines-Young, 2011).

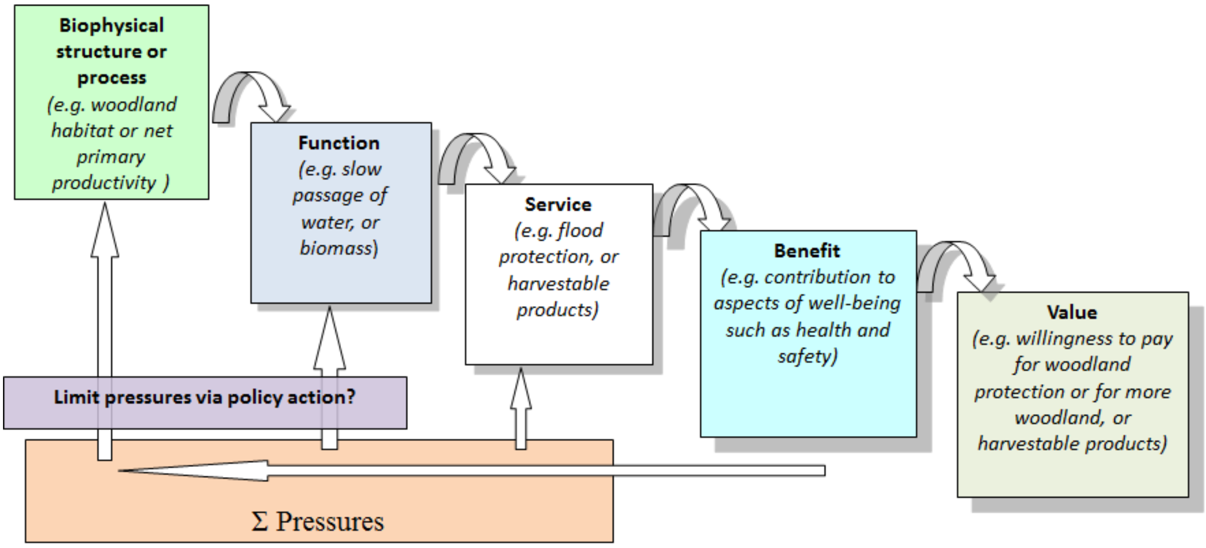

Figure 2: The ecosystem service cascade (Haines-Young & Potschin, 2011)

The fundamental principle of the ecosystem service paradigm remains: a human beneficiary needs to be identified for any ecosystem service to exist, thus it is essential to distinguish between 'final services' that contribute to human well-being and the 'intermediate ecosystem structures and functions' that give rise to them (Figure 2) (Balmford et al., 2008; COWI, 2014; Haines-Young & Potschin, 2014; Potschin & Haines-Young, 2011). This is done when attempting to complete a full ecosystem service assessment, including valuations, as to avoid double-counting (Boyd & Banzhaf, 2007; COWI, 2014; Fisher et al., 2009; Haines-Young & Potschin, 2014; MA, 2005; ONEMA, 2011; Potschin & Haines-Young, 2011; TEEB, 2010).

Considering the ecosystem service 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 (Brown et al., 2007; Costanza et al., 1997; Daily, 1997; De Groot et al., 2002; Potschin & Haines-Young, 2011). Jax (2005; 2010) emphasises the differences in the understanding of the term 'function' and the importance of choosing the appropriate definition for the task at hand. Hence, Wallace (2008) 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) further categorised the elements on the left-hand side of the ecosystem service 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 ecosystem services as three categories, namely, 'ecosystem benefits', 'core ecosystem processes' and 'beneficial ecosystem processes'. These are then ranked according to their contribution to human welfare. It is evident from the variety of approaches, including the ecosystem cascade, that the fundamental challenge is understanding the linkages between ecological systems and human well-being (Potschin & Haines-Young, 2011).

A further distinction needs to be made between biotic- and abiotic-dependent ecosystem services (COWI, 2014; Fisher et al., 2009; SEEA, 2012; TEEB, 2010). Some ecosystem service definitions consider ecosystem services to fundamentally depend on living organisms and/or processes. Therefore, all abiotic outputs from ecosystems are considered separately as part of 'natural capital' rather than ecosystem services themselves (Costanza, 2008; COWI, 2014). On the other hand, the French National Agency for Water and Aquatic Environments (ONEMA, 2011) differentiated, as previously mentioned, between 'environmental services' and 'ecological services', of which the former considers services as both biotic and abiotic compartments (COWI, 2014).

Figure 3 adapts the ecosystem service cascade model and 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., 2010b). This diagram graphically separates the elements of the cascade that are components of ecosystems and biodiversity, human well-being and 'final ecosystem services', as well as illustrating the impacts of feedbacks and management and/or restoration. Figure 3 introduces negative feedbacks into the conceptual model via environmental pressures. Mitigating policies includes more relevant features of real-world systems (Braat & De Groot, 2012; De Groot et al., 2010b). Positive feedbacks were included through institutions, judgements, restoration and management, thus effectively connecting social and environmental aspects of ecosystem services (Braat & De Groot, 2012). This adaptation addresses the issue of how the ecosystem service cascade model implies a 'unidirectional downward flow' from ecosystems to human well-being without any other forms of feedback or influences (Braat & De Groot, 2012; Salles, 2011).

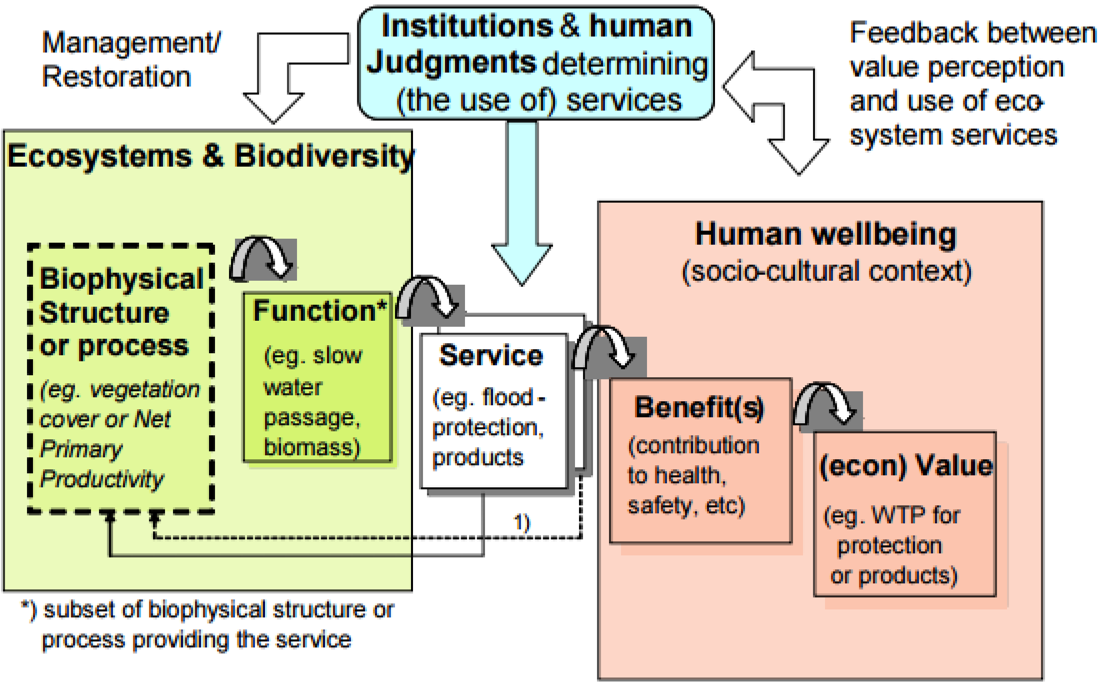

Figure 3: The pathway from ecosystem structure and processes to human well-being (De Groot et al., 2010b)

The challenge of identifying and classifying ecosystem services is conceptually and practically problematic as it requires a precise and commonly accepted definition of the term (Costanza, 2008; Haines-Young & Potschin, 2014; TEEB, 2010). This is because ecosystem services are essentially 'boundary objects'. They assist in transmitting and coordinating thinking between disciplines despite the absence of a benchmark definition (Costanza, 2008; Haines-Young & Potschin, 2014). The concept is widely used as it is vague and open to interpretation, thus any consensus regarding classification is difficult to realise (Haines-Young & Potschin, 2014). Ultimately, there are two primary definitional issues surrounding ecosystem services:

- Whether ecosystem services are considered as benefits or as contributions towards human well-being (through benefits supported by 'final' ecosystem services) (Costanza, 2008; Haines-Young & Potschin, 2014; Potschin & Haines-Young, 2011).
- Whether ecosystem services are only considered as outputs dependent on living processes, or if they include pure abiotic components (e.g. hydropower) (Haines-Young & Potschin, 2014).

2.2 Ecosystem Service Identification

Ecosystem services need to be described and measured before they can be valued, mapped or included in decision-making processes regarding the management of these services (Haines-Young & Potschin, 2013). Assessing ecosystem services is a three-step approach: identification, quantification and valuation (COWI, 2014; TEEB, 2010). The first phase of ecosystem service identification begins by analysing and comparing a list that classifies all relevant ecosystem services (e.g. Common International Classification of Ecosystem Services (CICES)) with available 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 facilitates 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 ecosystem services that includes an initial literature review and consultation of water management and ecosystem experts. This produced a qualitative description of the ecosystem services under study as well as a degree of categorisation. Such stakeholder interaction can often produce highly valuable information not available in the literature and which 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 the participatory process, the last step in the identification of ecosystem services may involve an on-site examination, additional mapping and/or local expert judgement (COWI, 2014). As ecosystem services are highly case- and site-specific, there is always the potential for a lack of data and information, thus not allowing for all ecosystem services to be accounted for (COWI, 2014; Saladin et al., 2012). Therefore, a step-wise process, even without a complete list of ecosystem services, may still provide some form of value in the communication and decision-making processes (Cardoso et al., 2013; ONEMA, 2011).

2.3 Ecosystem Service Classification and Typology

The classification and/or typology of ecosystem services is technically and conceptually challenging (Haines-Young & Potschin, 2014; Sokal, 1974). 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 & Potschin, 2014). Classification of ecosystem services can be broadly categorised into three main international classification systems, namely, the Millennium Ecosystem Assessment (2005), TEEB study (TEEB, 2010) and the CICES (COWI, 2014; Haines-Young & Potschin, 2013; MAES, 2012). These systems differ according to the definitions of ecosystem services and indicate how the ecosystem service approach has developed into a prominent tool for making environmental accounting protocols operational and detailing the values of ecosystem services to society and decision makers (COWI, 2014).

The Millennium Ecosystem Assessment (MA) 2005) popularised the concept of ecosystem services and basically classified ecosystem services into four broad categories, namely, provisioning, regulating, supporting and cultural services (Boyd & Banzhaf, 2007; Costanza, 2008; Fisher et al., 2009; Haines-Young & Potschin, 2014; ONEMA, 2011; SEEA, 2012; TEEB, 2010; Van den Berg et al., 2013; Van Wilgen et al., 2008). Figure 4 illustrates the various ecosystem goods and services categories, various drivers of change, and how these factors affect human well-being (CBD, 2008).

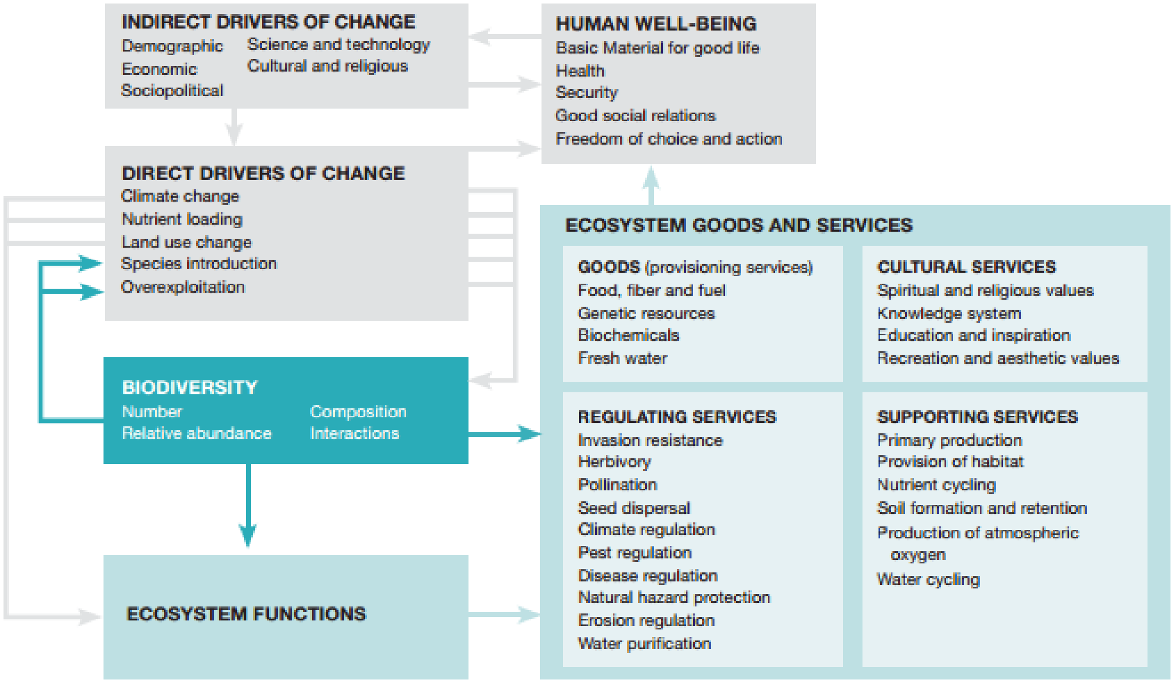

Figure 4: Biodiversity, ecosystem functioning and drivers of change (CBD, 2008)

Provisioning services can be described as the material or energy outputs from an ecosystem such as water (COWI, 2014; Haines-Young & Potschin, 2013; MA, 2005; TEEB, 2010). Consumers and buyers of these services can influence their production via different market mechanisms; governments can influence it via regulation (Van den Berg et al., 2013). Regulating or maintenance services have been described as the various ways biotic organisms moderate and/or mediate the surrounding environment, which include services that directly or indirectly impact human well-being (CBD, 2008; COWI, 2014; MA, 2005; SEEA, 2012; TEEB, 2010). 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; Ginsburg et al., 2010; MA, 2005; SEEA, 2012; TEEB, 2010; Van den Berg et al., 2013).

These three categories of ecosystem service are considered to fall under the overarching paradigm of 'final' ecosystem services. 'Final' ecosystem services are identified as contributions that ecosystems make towards human well-being (Balmford et al., 2008; COWI, 2014; De Groot et al., 2010; Haines-Young & Potschin, 2014; Potschin & Haines-Young, 2011). These ecosystem services always retain some form of connection to their underlying ecosystem functions, structures and processes that create them (COWI, 2014). Thus, the category of supporting services falls under the paradigm of 'intermediate' services that encompass all ecological/biological processes, structures and functions that underlie the provision of 'final' ecosystem services (Figure 2 and Figure 3) (Costanza, 2008; COWI, 2014; Haines-Young & Potschin, 2011).

The importance of ecosystem services and their classification lies in 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). Hence, Figure 5 illustrates the impact of direct and indirect drivers of change and ecosystem degradation as these have the potential to disrupt the natural flow of ecosystem services shown in Figure 2 and Figure 3.

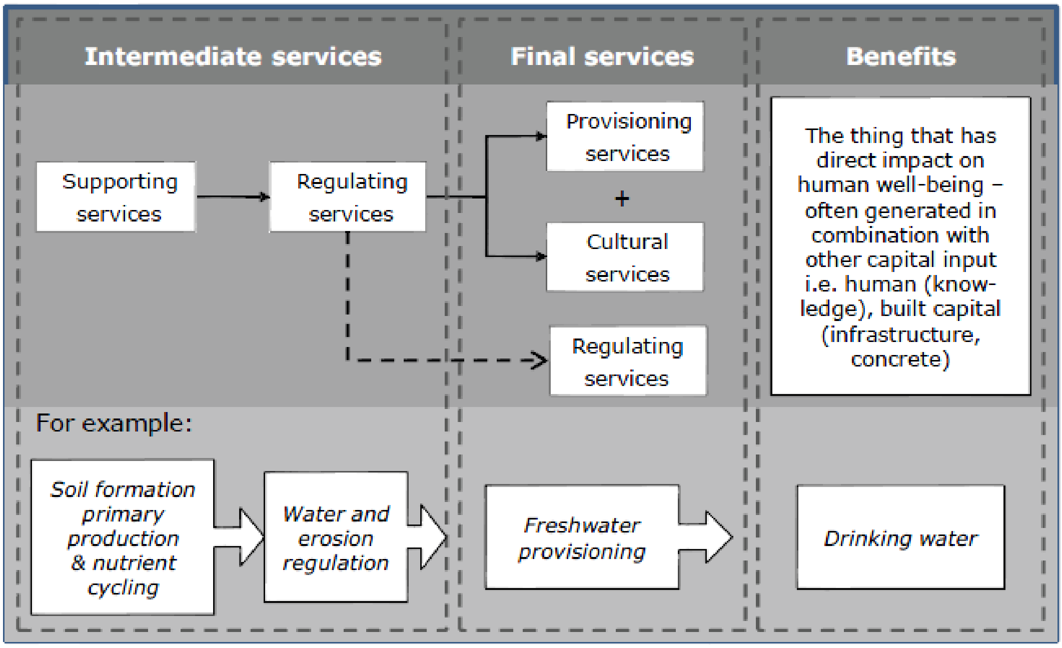

Figure 5: The distinction between intermediate services, final services and benefits (adapted from Fisher et al., 2008) illustrated by the stylised relationship between supporting, regulating, provisioning and cultural services (Ginsburg et al., 2010)

Figure 5 elucidates the relationship between the different MA (2005) categories of ecosystem services and intermediate services, final services and benefits by providing a basic drinking water example. The total economic value (TEV) framework (Section 2.3.2) directly aligns itself with the services and attributes framework, excluding supporting services (Turpie et al., 2010).

The CICES and TEEB classification schemes attempted to extend the MA (2005) approach for different applications. Table 1 compares the three classification schemes in terms of their individual ecosystem service categories. MAES (2012) describes how the MA classification is globally recognised and used for general sub-global assessments. The TEEB classification is based on the original MA classification and is used within current TEEB studies throughout Europe. The CICES scheme offers a hierarchal system tailored towards environmental accounting, which was developed on the foundations of the other two classification schemes.

Table 1: Ecosystem service categories in MA, TEEB and CICES (MAES, 2012)

MA categories	TEEB categories		CICES v. 4.3 group
Food (fodder)	Food	Provisioning services	Biomass (nutrition)
			Biomass (materials from plant, algae and animals for agricultural use)
Fresh water	Water		Water for (drinking purposes) (nutrition)
			Water (for non-drinking purposes) (materials)
Fibre, timber	Raw material		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)

MA categories	TEEB categories		CICES v. 4.3 group
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) Regulating and supporting services (MA) Regulating and maintenance services (CICES)	(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		(Mediation of) liquid flows
	Moderation of extreme events		
Erosion prevention	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			
Primary production Nutrient cycling (supporting services)	Maintenance of life cycles of migratory species (including nursery service)		Lifecycle maintenance, habitat and gene pool protection
			Soil formation and composition
			(Maintenance of) water conditions
	Maintenance of genetic diversity (especially in gene pool protection)		Lifecycle maintenance, habitat and gene pool protection
Spiritual and religious values	Spiritual experience	Cultural services	Spiritual and/or emblematic
Aesthetic values	Aesthetic experience		Intellectual and representational interactions
Cultural diversity	Inspiration for culture, art and design		Intellectual and representational interactions
			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)

Fisher et al. (2009) discuss several broad characteristics of ecosystem services that can assist with classifying ecosystem services for a variety of decision-making contexts. These are the public-private good aspect, spatial and temporal variation, joint production, complexity and benefit dependence. A thorough understanding of the key characteristics of ecosystem services can contribute towards developing and improving effective management approaches (Brauman et al., 2007; Fisher et al., 2009).

The public-private good aspect involves how different ecosystem services 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 something is considered a rival, then the use of that good/service 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 or user can keep others from using/consuming that particular good/or 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 how ecosystem services fit into the spectrum from excludable to non-excludable and rival to non-rival. Ecosystem services can fall anywhere within this spectrum; however, most ecosystem services are considered public goods. But, there are also many circumstances in which 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 ecosystem services fit into the public-private goods spectrum is complex as it is a function of individual ecosystem service dynamics as well as the various social systems that interact with these services (Fisher et al., 2009).

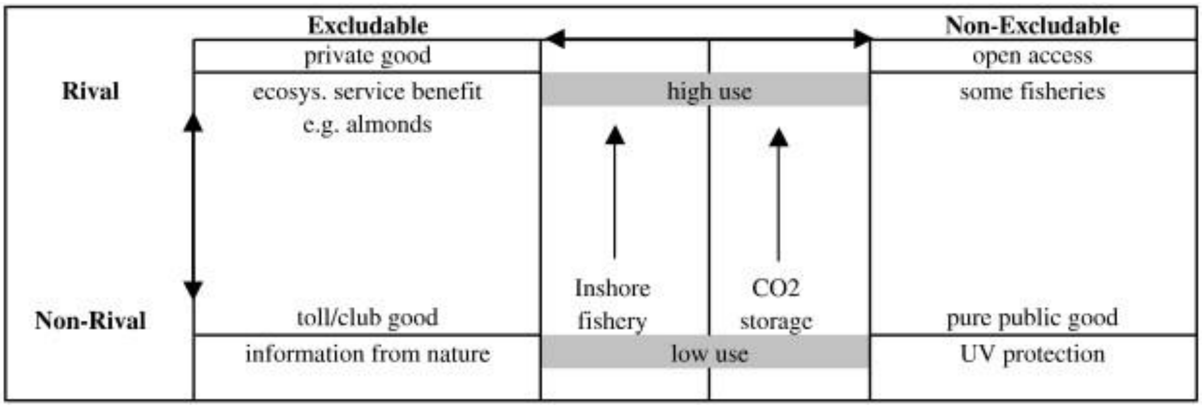

Figure 6: Good and service characterisation continuum from rival to non-rival and excludable to non-excludable (Fisher et al., 2009)

Ecosystem services are heterogeneous in space and time. Understanding this concept can assist classifying ecosystem services (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 downstream communities (Fisher et al., 2009). The concept of joint production is that ecosystem services can deliver multiple benefits for human well-being, and/or discrete ecosystems can provide several ecosystem services (Daily, 1997; Fisher et al., 2009). This characteristic of ecosystem services could be important for deriving classification schemes for specific 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 classifying ecosystem services as it allows for some degree of simplification and generalisation that can be tailored towards particular outcomes and applications (De Groot et al., 2010; Fisher et al., 2009; SEEA, 2012).

Boyd and Banzhaf (2007) outline the notion of benefit dependence: desired benefits dictate how an ecosystem service is understood. Whether an ecosystem service is considered as 'final' or 'intermediate' depends on the beneficiaries as well as what benefit is valued, monitored 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 the final service. Ultimately Fisher et al. (2009: 644) argue how 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 Managing Ecosystem Services Sustainably

The conception of ecosystem services and the abundance of scientific and socioeconomic 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 ecosystem services (Nahman et al., 2009).

Recent research has highlighted the widespread decline in ecosystem services as a result of their unsustainable use and human-driven ecosystem degradation (De Groot et al., 2010b; Egoh et al., 2008; Goodstein, 2011; MA, 2005; Nunes et al., 2014; WRI, 2001). Rockström et al. (2009) describe how humanity has transgressed at least three of nine planetary boundaries that define the safe environmental operating conditions for 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 natural resources and ecosystem services (De Groot et al., 2010a; O'Neill et al., 2010; Vihervaara et al., 2010). Thus, to secure the current and future provision of these crucial ecosystem services, areas that maintain the various ecosystem components, processes and functions need to be managed effectively and sustainably (Chan et al., 2006; Egoh et al., 2008; Fernald et al., 2012; Hanley et al., 2007; Van Jaarsveld et al., 2005).

De Groot et al. (2010b) and Gallopin (2003) highlight that most challenges facing the integration of ecosystem service management into conventional landscape planning, management and policy decision-making are due to the intrinsic complexity of socioeconomic systems and contrasting ideas surrounding the meaning and implications of sustainability (i.e. 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 as ecosystem services provide benefits directly and indirectly to people (DWA, 2013~~b~~; Fernald et al., 2012; Gallopin, 2003). This is closely linked to the notions of 'weak' and 'strong' sustainability. Hartwick (1978) and Solow (1993) introduce these concepts and describe how weak sustainability incorporates the idea that natural capital (i.e. stock of environmental assets) can substitute for human capital (i.e. labour, infrastructure etc.) while strong sustainability assumes that these two types of capital are complementary as opposed to being interchangeable (and therefore cannot substitute for one another).

Modern ecosystem service management is underpinned by the notion of sustainability; however, traditional (reductionist) scientific thought has not been able to cope with issues surrounding sustainability due to the complex and self-organising nature of these vulnerable systems (Hjorth & Bagheri, 2006; Limburg et al., 2002; Manuel-Navarrete et al., 2007; Nahman et al., 2009). Emerging schools of thought encourage systemic thinking (nonlinear and/organic) as a method for understanding the sources of and solutions to modern, complex problems such as ecosystem service decline (Hjorth & Bagheri, 2006; Palmer et al., 2004; Palmer et al., 2014). 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.

Payments for ecosystem services (PES) is a widely acknowledged and researched approach aimed at promoting the sustainability of ecosystem services through financial incentives. PES is defined by Wunder (2007) as “a voluntary, conditional agreement between at least one ‘seller’ and one ‘buyer’ over a well-defined ecosystem service – or a land use presumed to produce that service”.

PES agreements and policies attempt to compensate individuals or communities for engaging in specific activities that increase the provision of certain ecosystem services, which 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 ecosystem 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.

Wunder (2007) notes that the lack of a formalised definition of PES systems has led to much confusion around the topic. 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). Many people in southern Africa rely heavily on natural resources and ecosystems for their livelihoods. The overarching aim of PES schemes is to promote resilient livelihoods in the face of climate and disaster risks in an attempt to create win-win situations where key ecosystem services are restored so that ecosystem services become sustainable eventually (Mander et al., 2010; Midgley et al., 2012; Van den Berg et al., 2013). (Refer to Section 11.2 for more discussion on PES.)

This project considers sustainable utilisation as the use of ecosystem services at levels and in ways that allow them to continue renewing themselves indefinitely for all practical purposes. 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 the ability of an ecosystem 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 ecosystem service management into policy and decision-making requires a systemic approach with clear system specifications to which the concept of sustainability is applied (Gallopin, 2003; Limburg et al., 2002; Moffatt & Hanley, 2001).

2.5 Aquatic Ecosystem Services

Considering the previous discussion focusing on defining, identifying and classifying ecosystem services, it is critical to develop a clear and robust definition of AESs. Such a definition will allow for the selection of an appropriate classification scheme for this project, which will be able to address the decision context for which the AESs services are being considered. To address the primary research question and associated sub-goals of this project, final and intermediate AESs are defined within this study as:

Final AESs

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

Intermediate AESs

Biophysical functions, processes or conditions of aquatic ecosystems that only provide indirect benefits to humans by their 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 ecosystem services only include provisioning, cultural and certain regulating services that directly contribute towards human welfare. Intermediate ecosystem services could fall under any of the four ecosystem service categories outlined by the MA (2005). The concepts of ecosystem services are both considered to fall under the term ‘service’, and these individual concepts correlate with the notion of stocks and flows described above.

Aquatic ecosystem processes, functions and conditions include all abiotic and biotic components of ecosystem services as the definition of an ecosystem described by the CBD (1992) underpins the aquatic ecosystem component of this definition. As mentioned by ONEMA (2011) and SEEA (2012), this definition does 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. Ecosystem services derived from wetlands are excluded from this definition due to the methodical and conceptual issues related to delimiting aquatic ecosystems (especially wetlands), as explained by ONEMA (2011: 74). Wetlands are included in the causal analysis of the ecosystem and construction of the CLD as they play an integral role in the hydrodynamics and complexity of aquatic ecosystems.

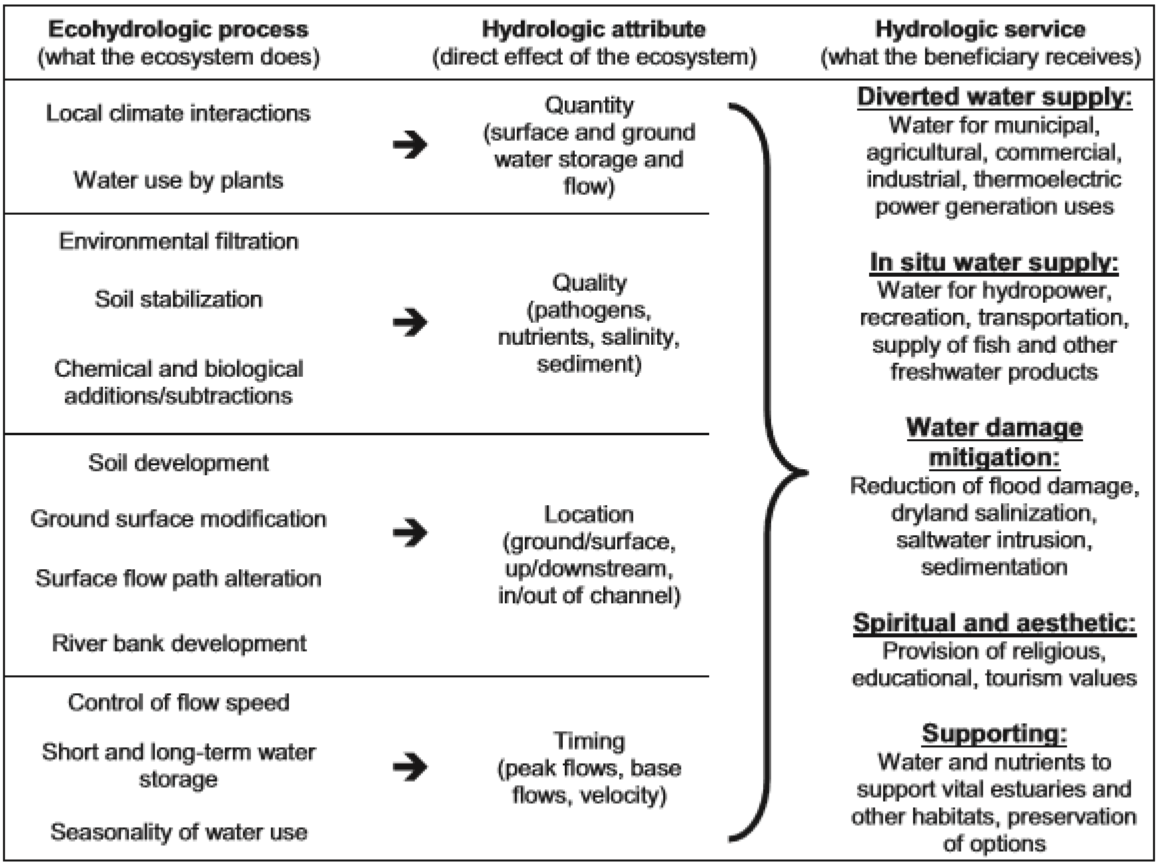

Figure 7: Relationship of hydrologic ecosystem processes to hydrologic services (Brauman et al., 2007)

Brauman et al. (2007) describe 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 ecohydrological processes (analogous to intermediate ecosystem services) can affect any of the four attributes. Understanding how intermediate and final AESs affect one another is incredibly important when trying to develop systemic models. These AESs are highly interdependent at different levels, thus understanding complex trade-offs between these services will help to identify whether the effects on one another are synergistic or competitive (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).

3 THE CONCEPT OF VALUE AND VALUATION OF AES

3.1 The Concept of Value and Valuation of Ecosystem Services

For the purpose of this project, 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 far beyond its intended scope (Parks & Gowdy, 2013). Since environmental goods and services (e.g. clean air and water) do not generally enter markets, or do so only imperfectly, market prices for these goods and services either do not exist, or capture their true value inadequately (Dixon & Pagiola, 1998). Thus, the monetary value of these benefits cannot be measured on the basis of existing market prices. In such cases, it is necessary to conduct an economic valuation exercise using a suitable non-market valuation technique. 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, air pollution (or, with a particular air pollution mitigation measure) or water filtration of wetlands.

Costanza et al. (1998) and Farber et al. (2002) emphasise the differences between economic and ecological understandings of value. They argue that due to the existence of potential win-win situations for human activities within the natural environment, valuation of ecosystem services 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). Turpie et al. (2010) define economic value as the most a person is willing to relinquish in terms of other goods and 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 illustrate economic value accurately as many individuals are willing to pay more than the required market price (Turpie et al., 2010).

Although flows of ecosystem services provide a nearly limitless set of valuable properties, a large proportion of their services remain unpriced through traditional markets (Hanley et al., 2007). Unfortunately, entirely inclusive valuations of ecosystem services have not been particularly successful due to a myriad of methodological challenges, and because it is not always possible to identify marketable value attributes of ecosystem services accurately (Barbier, 2011; Limburg, 2009; Parks & Gowdy, 2013). Thus, such goods and services are left without a market price; albeit not without value (Alpizar et al., 2007; Ferraro, 2000). Up until now, one way of accounting for such goods and services was to present them as intermediate goods and services to 'final' goods and services (i.e. goods and services that 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 distinction helps to prevent double-counting of intermediate and final goods and services as well as ecosystem functions (Boyd & Banzhaf, 2007; Fisher et al., 2009).

3.2 Valuation Methods for Ecosystem Services

There is a myriad of different approaches towards ecosystem goods and services valuation that include market value approaches (which involve the quantification of production), surrogate market or revealed preference approaches (which involve observation of related behaviour), and simulated market or stated preference approaches (which involve direct questioning) (Turpie et al., 2010). The TEV framework defines the types of ecosystem values to be quantified. However, there are numerous ways these values can be expressed that are entirely dependent on who requires the information and for what purpose the information is required (Turpie et al., 2010).

The TEV framework disaggregates the value generated by ecosystems into four categories, namely, consumptive or non-consumptive direct use value; indirect use value; option value; and non-use value (Goodstein, 2011; Hanley et al., 2007; Tietenberg & Lewis, 2010; Turpie et al., 2010). Value can be measured at local to national scales and from social or private perspectives. Environmental assets are often valued in terms of the nett economic benefits (value) they provide, which is often based on willingness to pay (WTP) estimates of intangible values (Hanley et al., 2007; Turpie et al., 2010). The notion of using nett present value requires one to apply a discount rate to valuing environmental assets and thus allows for the value of these goods and services to be analysed over periods of time (Goodstein, 2011; Hanley et al., 2007).

Pascual et al. (2010) argue that the value of ecosystems should account for two separate components from an economic standpoint. The first being a combined value of all ecosystem service benefits delivered in a given state, which is effectively the same as the notion of TEV (Pascual et al., 2010). The second component relates to the ecosystem's capacity to retain these benefits when faced with natural and anthropogenic variability and disturbance (Pascual et al., 2010). Balmford et al. (2008) and Gren et al. (1994) describe how the first aspect has often been referred to as 'output value' and the second aspect as 'insurance value'.

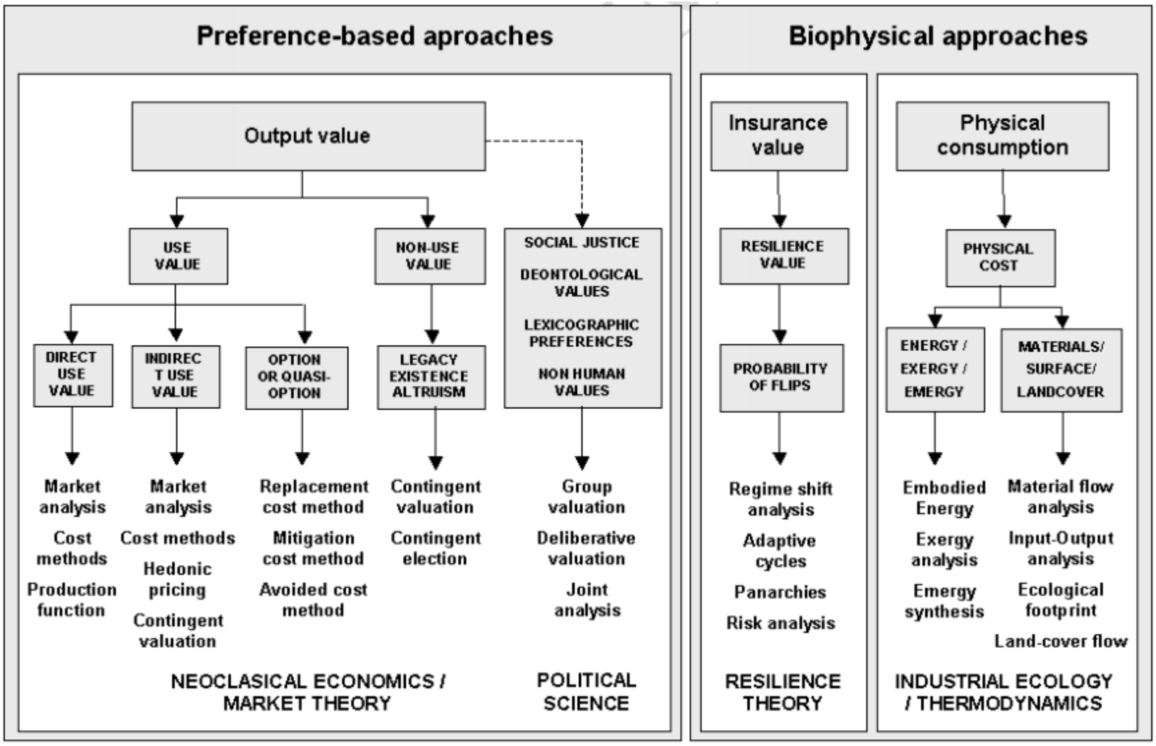

Figure 8: Approaches for the estimation of nature's value (Pascual et al., 2010)

Figure 8 illustrates many of the various approaches used to value natural assets such as ecosystem services. These methods have been separated into preference-based approaches and biophysical approaches alike (Pascual et al., 2010). However, for the purpose of this study, only the preference-based approaches are considered within the literature review as these fall under the neoclassical economics and political science groupings, which are the most relevant to the project.

Use values are based on actual physical use of environmental goods and services, whereas non-use values are not associated with actual use, or even the option to use an ecosystem and/or its services (Dziegielewska, 2013; Goodstein, 2011). Direct use values are derived through the consumptive or non-consumptive use of ecosystem services such as hunting, fishing, drinking water or hiking (Dziegielewska, 2013; Turpie et al., 2010). Indirect use values arise when ecosystems produce outputs that create inputs into separate production processes elsewhere (i.e. the benefits are realised off-site) (Goodstein, 2011; Turner et al., 1994; Turpie et al., 2010). For example, lower level organisms on an aquatic food chain can provide indirect use values to fishermen who catch the fish that prey on them (Dziegielewska, 2013).

Option value is the value placed on goods and services for their potential to be available in the future, even though it may currently not be used (Danielson & Leitch, 1986; Goodstein, 2011; Pascual et al., 2010). These values are very important when there is uncertainty regarding potential use and values of an ecosystem good or service in the future (Nhuan et al., 2003; Perman et al., 1996).

Existence values, which are considered non-use values, reflect benefits from one knowing that a particular good or service simply exists (Dziegielewska, 2013; Pascual et al., 2010). For example, many people would benefit or gain satisfaction from knowing that polar bears are alive regardless of whether they have ever seen one. Thus, many people are happy and willing to pay to protect this species' habitat (Turpie et al., 2010).

Bequest values make up the other non-use value component and specifically refers to the benefits attained from preserving particular goods and services for future generations (Goodstein, 2011; Turpie et al., 2010). For example, many people are willing to pay to reduce potential future damages because of climate change, which is despite the fact that most of these changes are predicted to occur long after the current generation is gone (Dziegielewska, 2013).

The concept of option values can be considered as use or non-use values. It is considered a use value as it assures direct or indirect use of the good or service in question, although it could also be considered as a non-use value as it is not related to a current use of the good and service. Barbier (1993) argues that it is difficult to determine future option values as they are very closely associated with present and future incomes as well as individual preferences. The theoretical concept of TEV is the sum of all above-mentioned values. However, irrespective of the methods of measuring them, they will not always be additive in nature (Turpie et al., 2010).

For many aquatic ecosystem products there are obvious, well-defined markets (e.g. fish), which make it relatively easy to estimate their value or worth (Barbier et al., 1997; Turpie et al., 2010). Though, often the prices or values of these goods and services are distorted; thus, they may not accurately reflect the social value of the goods and/or services (Perman et al., 1996). For example, it is difficult to value biodiversity or to quantify the aesthetic value of rivers to society.

Table 2 illustrates many of the commonly used natural resource valuation methods and the types of value that they are generally used to measure (Turpie et al., 2010). The potential number of methods that could be used to measure the different types of values decreases from left to right along the columns. Option values are rarely measured explicitly and are also incredibly difficult to separate from existence values in practice (Turpie et al., 2010).

Table 2: Commonly used natural resource valuation methods and the types of value which they are generally used to measure (XX = main use, X = possible use) (adapted from Turpie et al., 2010)

		Direct use value		Indirect use value	Option and non-use value
		Consumptive	Non-consumptive		
Market value approaches	Market valuation	XX	X		
	Production function	XX	X		
	Replacement cost/avoided damage	X	X	XX	
Surrogate market/revealed preference	Travel cost method	X	XX		
	Hedonic pricing	X	XX	XX	
Simulated market/stated preference	Contingent valuation	XX	XX	X	XX
	Conjoint valuation	X	X	X	X
Benefit transfer		XX	XX	XX	XX

According to Turpie et al. (2010), valuation methods can be separated in three overarching categories, namely, market value approaches, surrogate market approaches, and simulated market approaches (Table 2). These approaches are most commonly used to measure both direct and indirect use values of natural systems including ecosystem services. For the purposes of this study, these three approaches and the techniques of each approach will be briefly discussed. A discussion on the notion of PES is also included.

Pascual et al. (2010) distinguish between three main approaches to direct market valuation (or market value approaches): (1) market price-based approaches, (2) cost-based approaches and (3) production function based approaches. The primary advantage of these approaches is that they use data from existing markets and thus reflect real preferences and costs to individuals (Pascual et al., 2010).

Market valuation applies standard economic methods to value goods or services that are traded in formal markets. The particular types of costs and prices used are dependent on how one aims to express the value (i.e. economic surplus, nett private income, gross economic output or direct value added) (Turner et al., 2003; Turpie et al., 2010). Market price-based approaches are usually used to determine values of provisioning services because goods produced by provisioning services are generally traded on actual existing markets (i.e. agricultural markets) (Pascual et al., 2010). Using fisheries as an example, it should involve estimating a demand curve for fish and subsequently calculating the consumer surplus, then approximating total revenue received by fishermen, and subtracting the variable costs to estimated producer surplus (Turpie et al., 2010).

One can use surrogate prices for natural resources where there are no market prices. Barbier et al. (1997) suggest some possible methods for using surrogate prices. If the particular resource is traded or bartered, then it should be possible to derive its value from the market price of the commodity for which it is traded (e.g. fish for rice). Substitute prices can also be used if a close substitute for the good or service in question can be identified (Goodstein, 2011; Hanley et al., 2007). Delang (2006) suggests an alternative approach that includes estimating the amount of money people save by using natural products or ecosystem services as opposed to purchasing goods or services. It is also possible to deduce a minimum value for a good or service by estimating the value derived from the next best use (opportunity cost) of the inputs required for production or harvest of the good or service in question (Barbier et al., 1997; Turpie et al., 2010).

Indirect substitute prices use the opportunity cost of a substitute product as a proxy measure for the value of the good or service in question (Barbier et al., 1997). If one or more assumptions of the general equilibrium model are not met, which result in price or cost distortions in the market, then it may be necessary to adjust these prices and costs using shadow prices (Parks & Gowdy, 2013; Turpie et al., 2010). Shadow prices reflect economic value as opposed to financial value of particular goods or services as they are corrected to account for market distortions; their primary aim is to indicate the 'true', full value of a good or service to society (Parks & Gowdy, 2013; Turpie et al., 2010). Lastly, social survey methods, which require interviewing users, can be used as a market valuation approach when obtaining the necessary data, prices and costs (Ryan & Spash, 2011; Turpie et al., 2010). These interviews can be in the form of focus group discussions, key informant interviews or household questionnaires (Turpie et al., 2010).

Cost-based approaches are based on estimating the costs that would be incurred if ecosystem service benefits needed to be recreated through artificial means (Garrod & Willis, 1999). The main techniques associated with this approach are the avoided cost method, the replacement cost method and the restoration or mitigation cost method. The avoided cost method involves valuing the costs that would have been incurred in the absence of certain ecosystem services (Farber et al., 2002; Limburg, 2009; Pascual et al., 2010; Tietenberg & Lewis, 2010). The replacement cost method estimates the costs incurred by substituting specific ecosystem services with artificial technology (Farber et al., 2002; Goodstein, 2011; Limburg, 2009; Pascual et al., 2010). Lastly, the restoration or mitigation cost approach derives the costs of mitigating the impacts of loss of ecosystem services or the cost of restoring those particular services (Pascual et al., 2010; Turpie et al., 2010). This approach generally requires some form of probability analysis to determine the probability and extent of losses that could occur, thus assuming that loss or damage estimations are a measure of value (Turpie et al., 2010).

The production function approach advances traditional market valuation approaches by facilitating the estimation and inclusion of marginal values (change in value that would occur with a change in quality or quantity of ecosystem good or service) (Barbier, 1994; Ellis & Fisher, 1987). Pascual et al. (2010) describe the production function approach as estimating the amount a given ecosystem service contributes to the provision of another service or product that is traded through an existing market. The quantity of a good or service provided by an ecosystem is dependent on the attributes of the system itself and the inputs involved in the production of the good or service (Turpie et al., 2010). For example, the value of harvesting fish from a river is a function of the flow rate, water quality, availability of food, structure of the river etc. as well as of the labour inputs of fishermen. Therefore, any resulting improvements in the resource base or environmental quality derived from enriched ecosystem services will often lead to lower prices and costs as well as increases in the quantities of marketed goods, thus ultimately increasing consumer and producer surplus (Freeman, 1993; Pascual et al., 2010). The production function approach can be employed to value ecosystem services when there is sufficient scientific knowledge of ecosystem services that protect or support various economic activities (Barbier et al., 2009; Pascual et al., 2010).

Revealed preference or surrogate market approaches include two main methods, namely, the travel cost method and the hedonic pricing method. Revealed preference approaches are centred on observing individual's choices in existing, active markets that are directly related to the ecosystem service that is the subject of valuation (Pascual et al., 2010).

The travel cost method is based on the notion that recreational activities, such as visiting a game reserve, are associated with direct expenses as well as the opportunity cost of time. It thus attempts to determine the WTP for using an area from observed behaviour (Farber et al., 2002; Pascual et al., 2010; Turpie et al., 2010). The travel cost method assumes that the costs of a trip to a recreational site in terms of travel, entry fees, on-site expenditures and time can be used as a proxy for the use value of the site and for changes in its quality (Eshet et al., 2005; Eshet et al., 2006). The value of a change in the quality or quantity of a recreational site (e.g. changes in biodiversity) can be inferred by estimating the demand function for visiting the site in question (Kontoleon & Pascual, 2007).

The hedonic pricing method uses information about implicit demands for a particular environmental good or attribute of marketed commodities such as property (Farber et al., 2002; Pascual et al., 2010). It attempts to derive the contribution of environmental variables towards the value of certain properties using linear modelling of the various variables that make up property value (such as availability of water) (Turpie et al., 2010). The value of ecosystem services is reflected in property prices. Thus, a change in ecosystem services is reflected in the price of the property (Pascual et al., 2010; Turpie et al., 2010).

Stated preference approaches attempt to mimic a market and demand for a suite of ecosystem services by conducting surveys that address hypothetical, policy-induced changes in providing the specified services (Limburg, 2009; Pascual et al., 2010). These types of methods have been used to estimate the use and non-use values of ecosystem services, even in the absence of surrogate markets (Kontoleon & Pascual, 2007; Pascual et al., 2010). Turpie et al. (2010) argue that stated preference approaches should not be used to determine the value of ecosystem services as most people do not understand the complexity of ecosystem services and their linkages to economic activity. The main stated preference methods are contingent valuation, choice modelling or conjoint valuation, and benefits transfer.

The contingent valuation method (CVM) is a survey-based method where people are asked to state their WTP to receive a hypothetical benefit (e.g. an improvement in air quality), or to avoid a hypothetical loss. Or, conversely, their willingness to accept (WTA) compensation to forego a benefit or tolerate a loss (Eshet et al., 2005). It is called the CVM because the respondents' valuations are contingent or dependent on the hypothetical market setting established by the researcher. CVM is intuitive in principle, seemingly easy to apply and widely applicable to a range of different situations (since it is based on hypothetical scenarios such as hypothetical improvements or deterioration of the environment). However, the method is challenging to apply in practice, and the accuracy of the results is subject to debate. In particular, conducting a proper CVM survey that meets best-practice requirements is data-intensive, costly, and time-consuming (European Commission, 2013~~00~~). Nevertheless, the CVM method is one of few methods capable of estimating non-use values (Farber et al., 2002; Limburg, 2009; Pascual et al., 2010; Turner et al., 2003; Turpie et al., 2010).

The choice modelling or conjoint valuation method is a broad term for a variety of survey methods (e.g. choice experiments, contingent ranking/rating, paired comparisons etc.) that request respondents to rank/rate/choose alternatives rather than explicitly express a WTP or WTA. It was developed originally in the field of marketing, but is increasingly used to valuate ecosystem services (Carlsson et al., 2003; Stevens et al., 2000; Turpie et al., 2010). The most common approach attempts to model the decision-making processes of an individual within a specific context with the aim of estimating non-market values of ecosystem goods or services (Philip & MacMillan, 2005; Pascual et al., 2010). Each individual is faced with two or more alternatives to the good or service being valued, each with shared characteristics. However, each alternative has different amounts of each attribute, and one attribute is always the amount people would have to pay for said good or service (Pascual et al., 2010; Turpie et al., 2010). Thus, from the choices that people make between the alternative goods and services, a value for the chosen ecosystem good or service can be estimated. A monetary value is therefore obtained based on the trade-offs that respondents make between the monetary and non-monetary attributes. A baseline status quo alternative is usually included to help establish other alternatives in relation to the respondent's actual experience (Eshet et al., 2005).

Benefits transfer is an econometric tool for transferring existing estimates of non-market values (benefit and/or damage) from one study context to another, and making appropriate adjustments to account for differences in the two contexts (e.g. socioeconomic, demographic, geographic and climatic differences) (Georgiou et al., 1997; Barbier et al., 1997; Turpie et al., 2010). This method is generally accepted as a valid approach where time and budget constraints preclude primary valuation studies, provided that appropriate adjustments are made using statistical techniques. Turpie et al. (2010) note that the estimates of economic value at the study site is assumed to approximate the economic value of the good or service in question at a new similar site. This approach has been found to be significantly cheaper than alternative valuation techniques; however, many studies have rejected the accuracy and validity of this approach (Barton, 2002; Turpie et al., 2010).

3.3 Evaluating AESs in South Africa

Ginsburg et al. (2010) describe a simplified version of the South African Water Resource Commission's (WRC) framework for aquatic ecosystem service evaluation as illustrated in Figure 9. The four-phased approach includes (1) systems analysis, (2) assessing ecological change, (3) valuation of ecosystem services 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 ecosystem services (Ginsburg et al., 2010). The framework considers trade-offs between different water management scenarios by using economic valuations of the ecosystem services (Ginsburg et al., 2010). During the valuation of ecosystem services 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 ecosystem services 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 the valuation techniques are considered and attempts to statistically verify the results are made (Ginsburg et al., 2010).

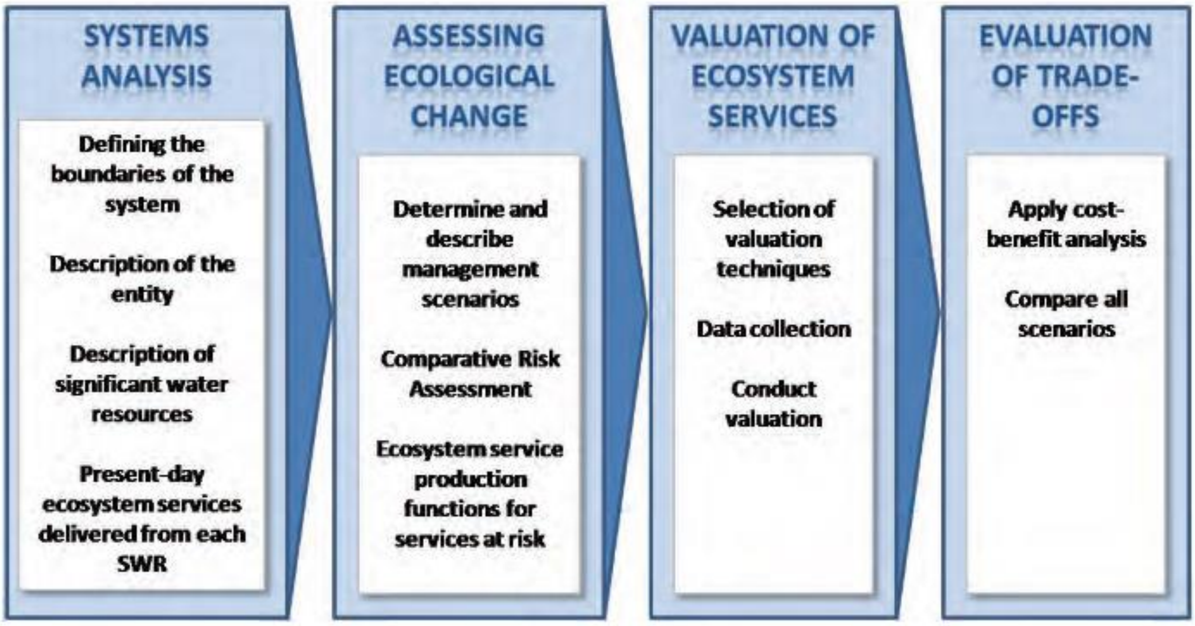

Figure 9: Simplified overview of the WRC aquatic ecosystem service evaluation framework (Ginsburg et al., 2010)

3.4 Challenges and Limitations of Ecosystem Service Valuation

There is a significant amount of literature regarding the issues and limitations attached to market and non-market valuation methods of ecosystem services and nature in general. Only general limitations and challenges of valuation approaches will be discussed. Analysing the limitations of individual valuation methods is beyond the scope of the project. The limitations associated with all types of ecosystem service valuation are primarily because there are no formal markets for most 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). In recent times, there has been much debate around 'best methods' to use when valuing ecosystem services, which is ultimately determined by the nature of the goods and services (Fisher et al., 2008; Van den Berg et al., 2013). The inherent trade-offs between costs and benefits of ecosystem services 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 the value of ecosystem services is due to several reasons. Firstly, 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, whereas they are accounted for in developed nations. The markets for these goods and services could distort their value due to external factors such as political decisions or media attention (Fisher et al., 2008; Limburg, 2009). Secondly, the complex nature of these goods and services cause them to be involved in multiple processes simultaneously, which makes it extremely difficult to analyse 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 (Limburg, 2009; 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.

Direct market valuation techniques are subject to significantly fewer limitations and criticisms than revealed and stated preference approaches. This is mainly due to the reliance of these approaches on cost or production data, which is usually simpler to obtain than the type of data required to establish a demand function for ecosystem services (Ellis & Fisher, 1987; Pascual et al., 2010). On the other hand, it is important to note that direct market valuation techniques are based on neoclassical economic assumptions and established market structures. The limitations associated with these assumptions are widely acknowledged and published (Goodstein, 2011; Hanley et al., 2007; Pascual et al., 2010). If there is no market for the goods and services in question, there will be no data available for these approaches. In addition, if the markets are distorted (i.e. due to subsidies or not being fully competitive), the value estimates will be biased (Pascual et al., 2010).

Revealed preference approaches are widely considered to be time-consuming and expensive due to the requirement for good quality and large data sets for every transaction as well as complex statistical analyses (Parks & Gowdy, 2013; Pascual et al., 2010). When faced with market imperfections and policy failures, which are common occurrences, revealed preference approaches can lead to distortions in monetary value estimations of ecosystem services (Pascual et al., 2010). Kontoleon & Pascual (2007) outline the main disadvantages of revealed preference approaches as being the dependence of value estimates on the technical assumptions made on the relationships between the ecosystem good or service and the surrogate market, as well as not being capable of estimating non-use values.

The hypothetical nature of the market in stated preference approaches has given rise to many queries concerning the validity of the estimates (Kontoleon & Pascual, 2007). A key question being whether the respondent's hypothetical answers to the hypothetical questions actually correspond to how they would react in real life (Pascual et al., 2010). Ryan and Spash (2011) discuss the varied nature of results from these methods due to there being no standardised survey and study design protocol. In theory, the concepts of WTP and WTA should be relatively similar in perfectly competitive markets; however, much of the literature indicates a strong divergence in value between the two concepts (Diamond, 1996; Garrod & Willis, 1999; Pascual et al., 2010). The 'insensitivity to scope' problem described by Kahneman (1986) is based on the notion that survey respondents do not consider the scope of environmental issues when deciding on their WTP except for particular scenarios. For example, people are often willing to pay similar (or even the same) amounts to prevent degradation of ecosystem services in a very small area (i.e. a river) as in a relatively large area (i.e. a drainage basin). Furthermore, Svedsäter (2003) argues how respondents' preferences for complex and unfamiliar ecosystem services are often not fully defined and thus stated preference approaches have the potential to render inaccurate results (Pascual et al., 2010).

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 understand the value of environmental goods and services without human agency being one of the dependent variables.

4 VALUE CHAIN ANALYSIS

4.1 Overview

A value chain analysis (VCA) 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 use, and discarding of residues (Baleta & Pegram, 2014; Kaplinsky & Morris, 2000). Lanen et al. (2008) define a value chain as the group of activities that converts 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. Mowen and Hansen (2011: 27) define the 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". VCAs are conceptual frameworks used to map and categorise chosen economic, social and environmental processes in service and product value chains. The ultimate aim is 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).

In general, most definitions involve the transformation of raw resources into some form of good and/or service (Baleta & Pegram, 2014; Baum, 2013). Service value chains as opposed to product value chains structure the value processes of a service firm or entity. The customer is considered throughput in the process (Bruhn & Georgi, 2006; Christensen et al., 2011~~03~~). On the other hand, traditional value chain frameworks tend to apply specifically to the throughput of material products (Baum, 2013; Kaplinsky & Morris, 2000). It is evident that the variability in application processes and potential analysis opportunities of VCAs are highly context specific, thus it is imperative to focus on the objectives and desired outcomes of the analysis.

The value chain is a type of workflow or template that defines internal processes or activities used by a firm. The value and cost of each of these activities signifies the approach used by the firm to implement its business level strategy (Kaplinsky & Morris, 2000). When the value chain can create additional value without incurring additional private costs, a competitive advantage is realised (Jordaan & Grové, 2008; Porter, 1985). Lynch (2003) argues 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.

Kaplinsky (2000) describes how significant developments in value chain frameworks have begun to provide an analytical structure that provides important insights into the two primary concerns with the determinants of global income distribution and the identification of effective policies to ameliorate trends towards unequalisation. Three important components of value chains need to be recognised to transform the approach from a heuristic tool into an analytical device (Kaplinsky, 2000). Firstly, value chains are repositories for rent (these rents are dynamic); 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, 2000).

Value chains are multi-faceted and can be locally, nationally and internationally oriented. They include activities such as cleaning, harvesting, design, transport, production, processing, transformation, marketing, packaging, support and distribution services, which are implemented by a variety of actors (Van den Berg et al., 2013). Value chains are often used to investigate governance, particularly the relationships, interactions and power between different chain actors (Humphrey & Schmitz, 2001). The 'governance' concept is central to global value chain approaches. It has been used to analyse the relationships and institutional mechanisms through which non-market coordination of activities within a chain take place (Humphrey & Schmitz, 2001; Stamm & Von Drachenfels, 2011).

According to USAID (2007), the structure of a value chain can be characterised in terms of five different elements. Firstly, the 'end market' is the starting point of the analysis and refers to the people who consume or purchase the product or service (Jordaan & Grové, 2008). The second element is the 'business and enabling environment' that 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 on its way 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. Horizontal linkages describe the cooperation between different firms that perform similar functions within the value chain itself. 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 & 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 for analysing many industries today (Peppard & Rylander, 2006). Thus, when considering the natural services provided by complex, nonlinear ecosystems that have limited physical and financial dimensions within their value chains, it is clear that traditional, linear, firm-focused VCAs cannot be used to analyse these chains.

4.2 VCA Approaches

In recent times, numerous different concepts have been used in other contexts that overlap with the concept of value chains. Thus, there is a myriad of different approaches towards VCA (Gereffi et al., 2001; Kaplinsky & 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 & Grové, 2008; Law, 1999; Peppard & Rylander, 2006). The general underlying concepts of each of these approaches are very similar; however, each has particular distinguishing characteristics.

Michael Porter introduces the term 'value chain' in his book "Competitive Advantage: Creating and Sustaining Superior Performance" (Porter, 1985). The VCA is based on Porter's generic value chain model that was developed with the aim of exploring Porter's model of competitive advantage through cost leadership strategies and differentiation (Baum, 2013; Porter, 1985).

Ketchen and Hult (2007) describe how Porter's value chain disaggregates a firm into its relevant strategic activities to understand the behaviour of costs and the existing and potential sources of differentiation (Porter, 1985). Hergert and Morris (1989: 183) state that "the fundamental notion in the VCA 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 & Hult, 2007).

Porter's competitive advantage model will not be discussed in this study as it is outside the scope of the project. Figure 10 illustrates Porter's classical value chain of a firm that 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 & Tejada, 2011).

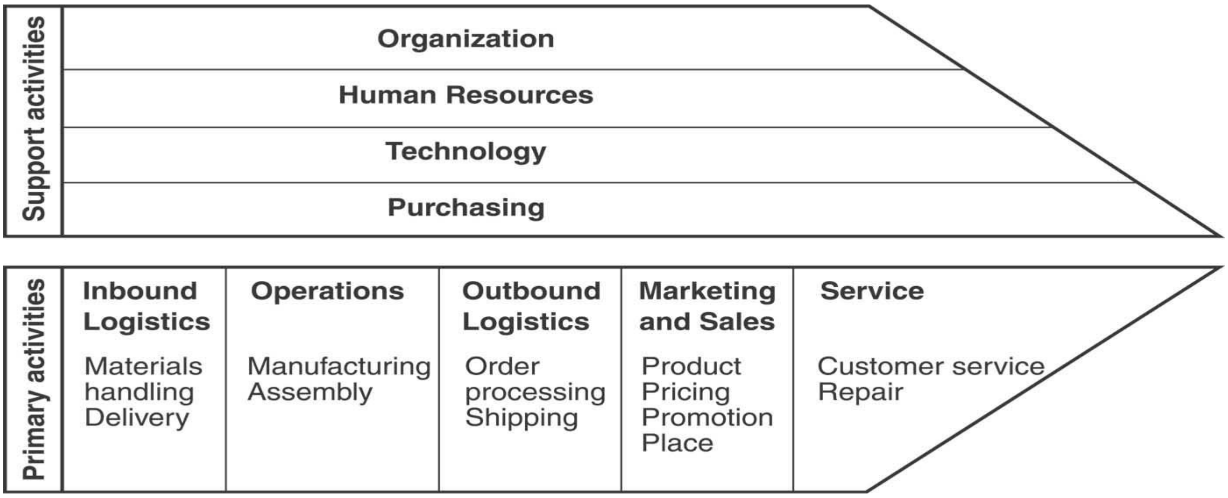

Figure 10: Porter's classical value chain of the firm (Porter, 1985)

Primary activities involve a product's physical creation, distribution and sales, and after-sales service (Mowen & Hansen, 2011), which 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 that describes the system of value chains within which a firm operates that contribute to moving the physical product to the end buyer. These systems can extend beyond the boundaries of an enterprise (Jordaan & Grové, 2008).

The approach to supply chain analyses has been used as a substitute to that of VCAs within similar contexts. Strictly speaking, 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) describe the term 'supply chain' as a generic label for an input-output based structure of value-adding activities that 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 & Grové, 2008).

The global commodity chain approach has also been used extensively as an alternative to traditional VCAs. The global commodity chain analysis has been primarily developed for individual commodity chains as opposed to entire value systems (Raikes et al., 2000). This approach emphasises the internal governance structure of supply chains (producer-driven vs buyer-driven) and the role of diverse lead firms in setting up global production and sourcing networks (Gereffi & Korzeniewicz, 1994; Gereffi et al., 2001). Its foremost contribution is the focus on the power relations embedded in VCAs.

On the other hand, the French *filière* approach has been described as a "loosely-knit set of studies with the common characteristic that they are a *filière* (or chain) of activities and exchange as a tool and to delimit the scope of their analysis" (Raikes et al., 2000: 13). Stamm (2004) explains that the primary objective of the *filière* approach is to identify the flow of goods and the actors involved within the flows, and to make them amenable towards economic analysis. According to Raikes et al. (2000), the *filière* approach is similar to Porter's concept of value chains based on its concern with quantitative technical relationships. However, the approach does not attempt to use any unified theoretical framework. Analysts conducting *filière* analyses rather borrow from different theories and methodologies (Jordaan & Grové, 2008). Kaplinsky and Morris (2000) highlight a key criticism of the *filière* approach in that it is static by nature, which implies it can only reflect relations at a specific point in time. However, Stamm (2004) maintains that this particular limitation has been overcome in recent times as it has been further developed through connections to numerous strands of theory. This approach indicates that it is not completely necessary to select only one method for analysing a value chain, but that it may rather be worthwhile to consider particular useful features from all other approaches that will contribute to a more in-depth analysis (Jordaan & Grové, 2008).

Henderson et al. (2002) introduce the concept of global production networks, which is a direct enhancement of the global commodity chain. A chain plots a vertical sequence of events leading to the delivery, consumption and maintenance of particular goods and services. It recognises that different value chains often share common economic components and are dynamic in that they are continuously reused and reconfigured. As such, a network focuses on the nature and extent of the interfirm relationships that combine groups of firms into larger economic groupings (Sturgeon, 2001). The major adjustment made to the global commodity chain is that the interrelations between the links of the chain and where they are embedded are now the focus of interest (Henderson et al., 2002). The difference between producer-driven and buyer-driven value chains is not retained within the global production network approach (Jordaan & Grové, 2008). Network approaches emphasise a major weakness of the chain-based approaches: the conceptualisation of production and distribution processes is essentially vertical and linear (Henderson et al., 2002; Jordaan & Grové, 2008). Such processes are better understood as highly complex network structures with intricate links creating multidimensional, multilayered frameworks of economic activity (Henderson et al., 2002).

Bolwig et al. (2010) introduce a conceptual framework aimed at overcoming many limitations of traditional stand-alone value chains through integrating specific vertical and horizontal components of value chains that together affect poverty and sustainability. The conceptual framework considers local and global environmental concerns but does not address ecosystem services specifically. It focuses on the nature and function of various actors within the value chain and 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 VCA can be loosely compared to the process and function of CLDs as positive and negative impacts, which 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 actor-network theory (ANT) approach emphasises the rationality of both objects and agency in heterogeneous networks, indicating that entities within particular networks are shaped by, and can only be understood through, their connectivity and relations to other entities (Law, 1999). Relating to the study of global production networks, this means that space and distance must be viewed as 'spatial fields' and relational scopes of power, influence, and connectivity rather than in absolute, Euclidean terms (Harvey, 1969; Murdoch, 1998). The ANT considers networks as hybrid collectives of human and non-human elements that allows for the consideration of significant technological elements that influence different economic activities (Henderson et al., 2002).

Peppard and Rylander (2006) developed the network value analysis (NVA) approach that aims to generate comprehensive descriptions of where value lies in a particular network and how value is created. An NVA approach involves five steps: 1) Define the network, 2) Identify and define network entities, 3) Define the value each entity perceives from being a network member, 4) Identify and map network influences, and 5) Analyse and shape. The NVA approach was developed as a method to analyse competitive ecosystems and provide a more holistic view of the complex interactions between individual components (Peppard & Rylander, 2006).

Lastly, the commodity chain approach forms part of larger set of different approaches to chain analysis, including value chains and global commodity chains. Tallec and Bockel (2005) describe how commodity chain analyses are specifically targeted at agricultural commodities, which differentiates them from the above-mentioned approaches. The commodity chain approach allows one to identify relationships between various different stages of transformation within the network of agricultural or agro-food systems (Jordaan & Grové, 2008). A commodity chain can be defined as a succession of operations and agents which, beginning upstream with raw materials, eventually emerge downstream, after numerous stages of transformation and increases in value, with one or more final products at the consumer level (Tallec & Bockel, 2005).

4.3 Limitations Surrounding VCA

Focusing on real-world value chains, products move through corresponding links, which may form part of alternate value chains. Thus, these chains are generally more complex than the definition of a simple value chain (Kaplinsky & Morris, 2000). Traditional VCAs only apply to products and services for which there are active markets; thus, specific ‘values’ of goods and services can be deduced. Ecosystem services that are not traded in formal or informal markets are therefore excluded from this form of traditional analysis. However, a multitude of techniques such as PES have been developed as an attempt to identify and present tradable value attributes of ecosystem services that could allow for some form of VCA (Brauman et al., 2007). Products and services embody 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).

Several approaches to VCAs have been developed in response to a variety of challenges and suitability factors associated with individual methods and applications. Within VCA, there is a proliferation of overlapping concepts and names. This makes 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 a common ground much greater than their distinguishing features. Porter (1985) describes the traditional value systems framework as a set of interlinked ‘complete’ entities that have all necessary business functions. Gereffi et al. (2001) argue that an advantageous point of departure from this notion, used by many other approaches, is to allow for ‘incomplete’ entities or firms that have specialised in certain value chain functions. Jordaan and Grové (2008), and Ramsay (2005) argue how the concept of a value chain could be considered 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 & Grové, 2008).

There has been a significant increase in the number of VCA case studies; however, many of the concepts surrounding the approach are still not very clear (Gereffi et al., 2001). Without a well-defined theoretical framework, there are severe limits on both the outcomes that can be derived from diverse case studies and comparisons between different value chains (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 VCAs 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 supporting complex natural environment require that we reconsider traditional methods for analysing complex and competitive environments (Peppard & Rylander, 2006).

Traditionally, VCAs are geared towards linear processes and private goods that form part of a conventional neoclassical market system. Thus, the notion of incorporating public goods such as ecosystem services, which generally do not have defined market values, into a VCA requires an alternative approach to the conventional linear techniques (Henderson et al., 2002; Van den Berg et al., 2013).

4.4 Ecosystem Services and Value Chains

Traditional VCAs only apply to products and services for which there are active markets. Thus, specific 'values' of goods and services can be deduced. Ecosystem services that are not traded in formal or informal markets are therefore excluded from this form of traditional analysis. However, the notion of payments for ecosystems goods and services (Brauman et al., 2007) was an effort to identify and present tradable value attributes of ecosystem services that will allow some form of VCA. Unfortunately, the inclusive valuation of ecosystem services has not been particularly successful (Barbier, 2011) because of methodological challenges (Parks & Gowdy, 2013). Furthermore, it is not always possible to identify marketable value attributes of ecosystem services, which then leave such goods and services without a market value, albeit not without value (Alpizar et al., 2007, Ferraro, 2000). Until now, one way of accounting for such goods and services was to present them as intermediate goods and services to 'final' goods and services (i.e. goods and services that does 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.

However, current controversies in applying valuation techniques derived from a narrow interpretation of value has 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 & Spash, 2011). This while the fields of behavioural psychology, neuroscience and social anthropology have shown that the human decision-making process is also a social and not only an individual process (Dawnay & Shah, 2005; Fischer & Hanley, 2007; 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 environmental goods and services.

Although economic systems and environmental ecosystems share many attributes, valuation of ecosystem services has traditionally been driven by short-term human preferences (Limburg et al., 2002). Limburg et al. (2002) argue how the valuation of ecosystem services needs to switch from choosing among environmental resources to valuing the avoidance of dramatic ecosystem changes. 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 goods and services have to be analysed within their individual spatiotemporal context. The application of a narrow interpretation of value does not present the full realm of value as presented by ecosystems. There is a growing acceptance that neoclassical welfare economics is limited (Goodstein, 2008; O'Neill et al., 2010) in reflecting this value. Alternative approaches include behavioural economics (Mullainathan & Thaler, 2000), nonlinear complexity theory, evolutionary economics, new institutional economics, post-Keynesian economics, and neuro-economics. However, ecological and environmental economists in South Africa have been slow to take advantage of these new developments (Nahman et al., 2009). Given the above-mentioned difficulties, we cannot follow a traditional approach to VCA (Kaplinsky & Morris, 2000).

Incorporating ecosystem services thinking into value chain assessments is a relatively recent consideration and thus literature is scarce (Van den Berg et al., 2013). Many ecosystem services have been directly or indirectly addressed through approaches to increase the sustainability of value chains, which include certification schemes, corporate social responsibility, risk management and mitigation initiatives (Grigg et al., 2009; Weiss et al., 2011). Christensen et al. (2011) demonstrate a combined economic and ecological approach aimed at emphasising both disciplines equally. The study focuses 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 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 International Union for the Conservation of Nature and Natural Resources' Global Business and Biodiversity Program (Bishop et al., 2008), the European Union Business and Biodiversity platform, the UNDP protecting biodiversity in working with agribusiness project (Leibel, 2012), and the Business and Biodiversity Offsets Program (Van den Berg et al., 2013). These initiatives and research endeavours emphasise the limits of market-based approaches for value chains that range from unorganised and powerless workers and the lack of true market values for ecosystem services 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) describe a case study exploring the governance options available to the Dutch government to promote the sustainable use of ecosystem services within selected tropical timber value chains and how ecosystem services can be given a more explicit place in the public and market mechanisms that govern tropical timber value chains. The study concludes that in most circumstances, the term 'ecosystem services' has not yet been defined accurately and clearly, 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 for developing a suitable tool that can conduct a VCA for ecosystem services. This is even though the timber value chain can be assessed using traditional value chain approaches as there is a reliable market value for the goods in question.

Stamm and Von Drachenfels (2011) emphasise how VCA generally begins 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 affect, and be affected by other peripheral value chains of associated goods and services, be it directly or indirectly. The notion of incorporating systems thinking and complexity theory into VCA links directly with the concept of ecosystem services, which is a product of highly complex and dynamic systems (Brauman et al., 2007; Limburg, 2009). The statement by Hergert and Morris (1989: 183) about how the main concept in VCA is that a particular product or service will increase in value as it moves through various processes required to develop it into a final good or service. On the surface, this statement seems to hold true for complex ecosystem services as there are numerous underlying ecological processes and functions that increase the value of these goods and services over time until they are consumed/used (Fisher et al., 2009; Johnston & Russell, 2011; Landers & Nahlik, 2013).

This project assesses the suitability of using CLD as an alternative VCA tool. We used a transdisciplinary approach to construct CLDs for each ecosystem good or service under consideration (Finnoff et al., 2005; Schaffernicht, 2010; Simonovic and Fahmy, 1999; Stewart et al., 1997). We hosted workshop(s) where experts from different disciplines helped to construct the diagram around the requirements of the good or services and to describe the services.

The fundamentals of CLDs are described by Sterman (2000: 137-190). Conceptualising a complex ecological system not only requires a clear definition of the key elements of the system and the causal relationships between these elements, but also an account of the relationship of the system with other systems (including social and economic systems). Here, ‘complexity’ refers to the emergent properties of the ecosystem that cannot be described in terms of the separate components of the system, but rather in terms of the interactions between system components (ecosystem services often fall in this category). Insight in 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 does exactly that, which then generates a mutual understanding and improved insight among stakeholders on how the system works and why it responds to external stimuli the way it does. Such insight is immensely powerful for strategizing ecosystem management challenges because it makes critical elements which are within the stakeholder control explicit, which allows scenario building, scenario analysis and making projections which allows the formulation of ecosystem management strategies in a proactive way.

5 SYSTEM DYNAMICS, CLDs AND VCAs

5.1 System Dynamics

Nonlinearities and temporal and spatial lags are common features of natural systems; however, these system characteristics are far too often not considered in scientific and economic investigations (Costanza & Ruth, 1998). Consequently, this reduces the ability of these inquiries to provide accurate insights necessary to make appropriate management and policy decisions to improve resulting system behaviour and performance for complex socioecological systems (Lane, 2008). One area of systems thinking is called system dynamics and is designed to specifically address these system features.

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 system’s behaviour (Kirkwood, 2013; Richardson & Pugh, 1989; Sterman, 2000). It is an interactive approach that can be used to explain intricate ecosystem structure and functionality and illustrates the outcomes of potential management strategies. 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 & Bagheri, 2006). Here, the term ‘complex’ refers to the emergent properties of the ecosystem that cannot be described in terms of the separate components of the system, but rather in terms of the interactions between system components (ecosystem services often fall into this category).

The original concepts and methodology were developed by Forrester (1961). It has been refined and improved over the past five decades. Initially only applied to business and industrial systems management, the scope and use of system dynamics has expanded to other focus areas including complex socioecological systems (Costanza & Ruth, 1998; Hjorth & Bagheri, 2006; Kelly, 1998; Kirkwood, 2013; Ruth, 1993). System dynamics modelling is an approach that attempts to close temporal and spatial gaps in decision-making, actions and effects (Arquitt & Johnstone, 2008; Costanza & Ruth, 1998). Different behavioural modes of complex systems are caused by changes in dominance between different feedback loops, most of which involve nonlinearities, delays, and accumulation and draining effects (Lane, 2008). 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 & Bagheri, 2006; Richardson, 1994).

System dynamics-based studies begin with conceptualising 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 & Ford, 1998; Doyle & 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 several computer programming tools and languages (e.g. Vensim and STELLA) that 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 & Ruth, 1998).

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 the internal structure of a system as more important than external events regarding how a problem or change comes about (Kirkwood, 2013). System dynamics diagramming methods have changed significantly over time, or have been altered in their use (often by software developments). However, there is little empirical proof of the efficacy of these different methods (Lane, 2008).

Figure 11 illustrates three different viewpoints of cause and effect. System dynamics considers all three viewpoints but focuses on nonlinear feedbacks

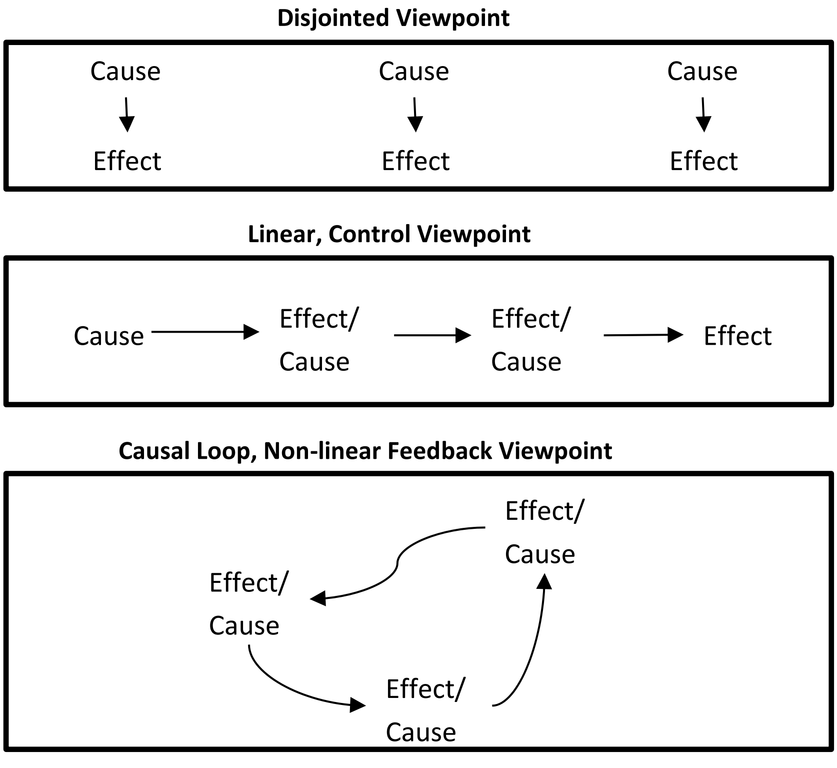

Figure 11: Three viewpoints of cause and effect (Hitchins, 2005)

The field of system dynamics uses two overarching methods for solving complex problems: qualitative and quantitative modelling approaches (Hürlimann, 2009). This project focuses on qualitative modelling in the form of CLDs and stock/flow diagrams. 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, it is clear that they are best used in conjunction with one another (Hürlimann, 2009; Lane, 2008; 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 as it allows for communicating information about the structure of the system visually and without using technical jargon while triggering discussion and debate among participants to create a mutual understanding regarding the system at hand (Stave, 2003; Vennix et al., 1997). Such a mutual understanding is important to create buy-in and to reach consensus regarding the significance of the goods and services derived from the ecosystem (Stave, 2003).

The process of creating a system dynamics model not only creates buy-in from participating stakeholders, but also makes the underlying assumptions of the model explicit. Thus, the model can be used to simulate the effect of ecosystem management actions and/or external disturbances on the system as a whole. When system dynamics models are used interactively in a public forum, they have the potential to illustrate resource systems and demonstrate different outcomes of strategies proposed by participants and/or managers (Stave, 2003; Vennix et al., 1997). Forrester (1987) emphasises that system dynamics modelling is necessary because while people can observe the structure of a system, they are unable to predict how causal relationships will balance in the system.

Communication is the primary role of diagrams in system dynamics. Diagrams 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). Model conceptualisation stimulates and frames MMODSs in such a way that facilitates the use of system dynamics. Diagrams are evolving thinking tools that characterise an individual or group’s understanding of a particular issue and communicates their assumptions unambiguously, which eventually leads to creating a fully formulated simulation model (Lane, 2008; Randers, 1980). Model exposition represents the assumptions that underpin mathematical models (Lane, 2008).

Figure 12 uses a bathtub analogy to illustrate the different types of variables and causal links used in system dynamics.

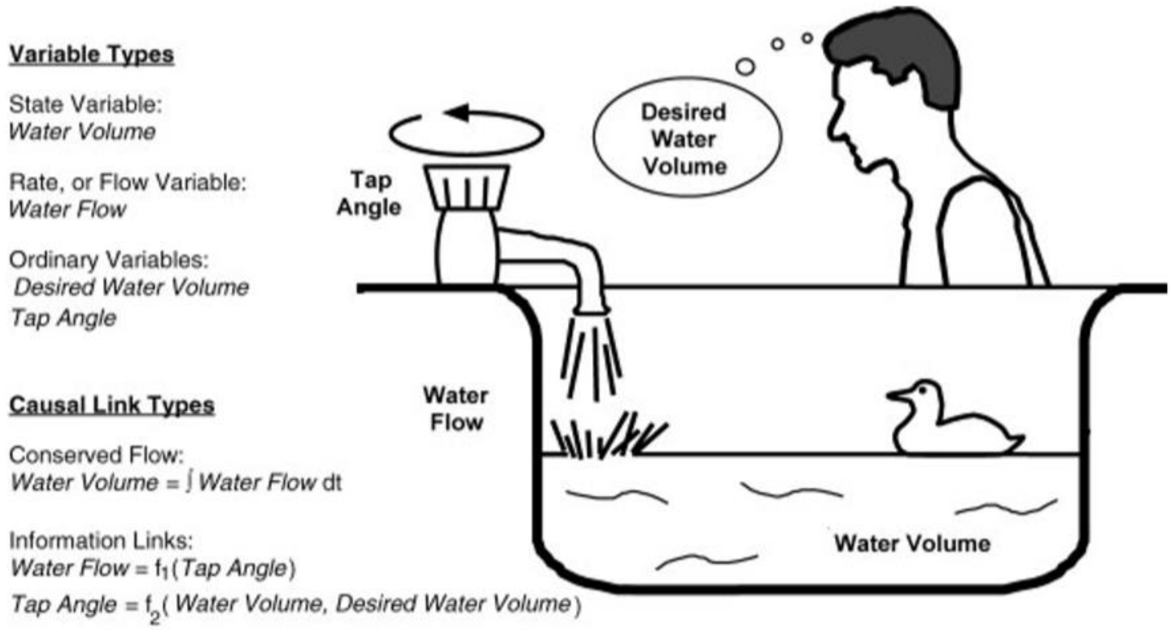

Figure 12: Illustration of variable types and causal links in system dynamics (Lane, 2008)

There are three types of variables and two types of causal links in system dynamics. 'State variables', 'stocks' or 'levels' (i.e. water volume) are only changed by accumulation/drainage in/out of them. 'Rate' or 'flow' variables (i.e. water flow) are the quantities or magnitude of those accumulating or draining processes (Lane, 2008). 'Ordinary' or 'auxiliary' (i.e. 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 the stock/flow diagram (SFD) are considered the leading qualitative modelling diagramming conventions overwhelmingly accepted by the international 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 13 illustrates the bathtub filling example (Figure 12) using a CLD and an 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 will be discussed in the next section. Ford (1999: 14-24), Sterman (2000: 191-262) and Lane (2008: 10-11) discuss these components of SFDs in more detail.

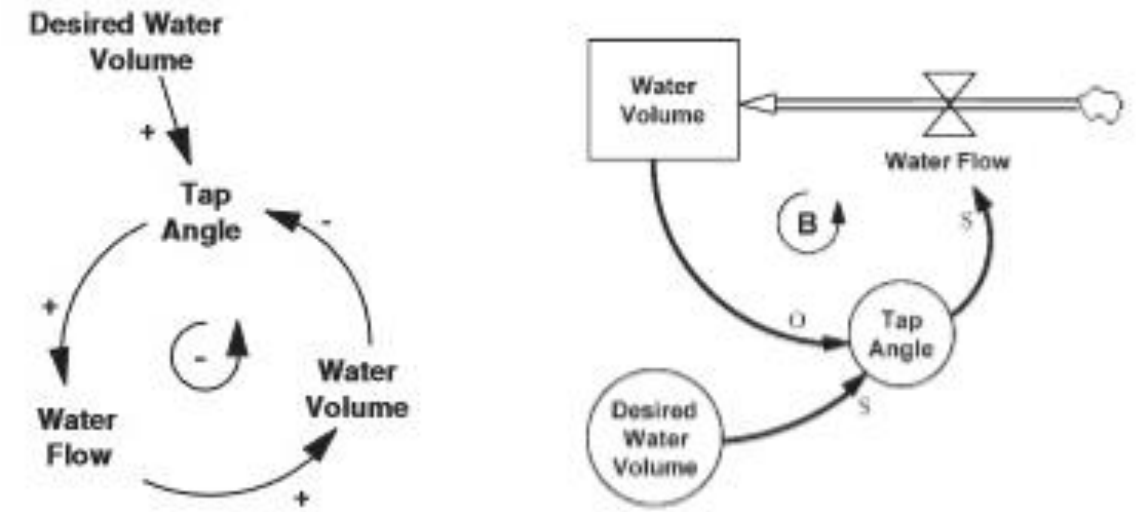

Figure 13: CLD (left) and SFD (right) illustrating the bathtub filling example (Figure 12) (Lane, 2008)

A system dynamics analysis proceeds through several major steps: problem definition and delimiting, description of the underlying system, model development, model verification, modelling for analysis, and communication (Ford, 1999; Richardson & Pugh, 1989). Participants and stakeholders in the modelling process can be involved at different stages of the process. On one extreme, it can be used simply for communicating the results of a completed model and for illustrating the effects of alternative options (Hale, 1993). However, a more participatory application will (where applicable) incorporate the participant's suggestions, enhance participant understanding of the resource system, and ultimately improve consensus building and communication.

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

5.2 CLDs

5.2.1 Causal loop theory

CLDs – also referred to as 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, to quantify the interactions between elements, and to incorporate accompanying time delays in the system (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 mutual understanding and improved insight among stakeholders regarding 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 strategizing ecosystem management practices because it makes critical elements, which are within the stakeholder's control, explicit. This increases scenario building, scenario analysis and system projections, thus potentially allowing for the proactive formulation of ecosystem management strategies.

The reason for often 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 focusing on the trade-offs between environmental conservation and socioeconomic benefits (Evans, 2004). These challenges are derived mainly from difficulty in communicating the value of ecosystem services and communicating the complexity of the underlying ecosystems to a broad audience consisting of different technical backgrounds and potentially conflicting perspectives (Costanza & Ruth, 1998; Lane, 2008; Morecroft, 1982; Sterman, 2000). Such support requires improved insight into the value of ecosystem services through an improved understanding of the workings of the ecosystem itself.

A typical CLD comprises 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 that 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; Schaffernicht, 2010; Sterman, 2000). 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; all other variables need to be held constant (*ceteris paribus*) (Ford, 1999; Richardson, 1986; Schaffernicht, 2010; Sterman, 2000). This doctrine allows one to analyse the impact of one change on the system at a time, which increases the transparency of system analyses.

Each arrow is labelled with either a + or a − that 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. 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 & Pugh, 1989).

The most relevant loops are highlighted by 'a loop identifier' that determines whether the loop is negative (balancing) or positive (reinforcing) (Sterman, 2000). Negative feedback loops act to negate outside disturbances. 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 system management (Fernald et al., 2012; Ford, 1999; Kirkwood, 2013). Figure 14 is an example of a CLD that includes independent variables (e.g. run-off), dependent variables (e.g. stream flow) and arrows indicating causal links.

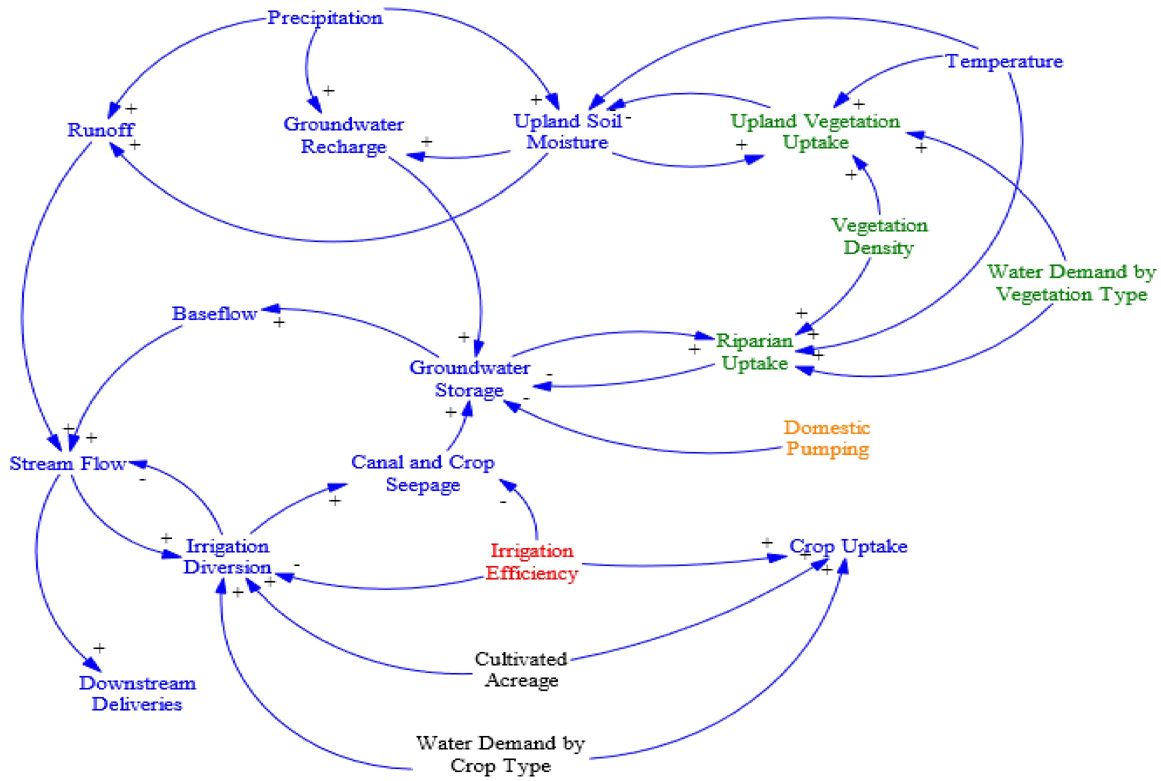

Figure 14: Hydrology subsystem CLD (Fernald et al., 2012)

5.2.2 CLDs and complex systems

Costanza and Ruth (1998), Evans (2004), Ford (1999), Kirkwood (2013), and Sterman (2000) explain 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 & Ruth, 1998). Stocks are variables in a system that exist at a point in time, for example, the amount of water in a dam (Evans, 2004).

These stocks are in turn influenced by controls or flow variables (Costanza & 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), for example, the amount of water abstraction 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 & 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).

Armah et al. (2010) demonstrate an example of how qualitative modelling – CLDs specifically – can be used to conceptualise complex socioeconomic systems and assist in developing strategies to improve management and decision-making within the system. The study explored the impact of flooding on rural communities in northern Ghana~~Nigeria~~ that are dependent on the supply of crucial ecosystem services and natural resources. CLDs were used to develop and analyse flood-induced coping schemes. This enabled the intrinsic characteristics of the socioeconomic system under study, which appeared to reduce vulnerability and mitigate flood damage risk, to be identified (Armah et al., 2010). Figure 15 demonstrates one of the CLDs used to analyse the system.

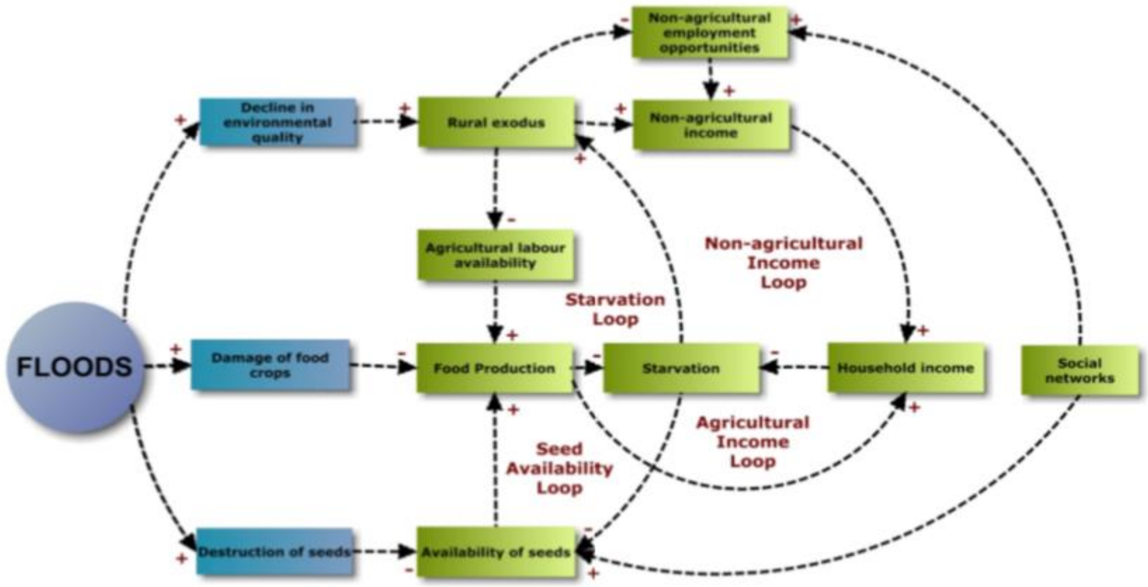

Figure 15: Simplified CLD illustrating the impact of floods on communities in northern Ghana (Armah et al., 2010)

Pollard et al. (2008), Pollard et al. (2013), and Pollard et al. (2014) outline a few scenarios where CLDs, or similar types of qualitative modelling tools, were used to map different components of a complex socioeconomic 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 ecosystem services (Figure 16). However, ecosystem services were considered as a whole as opposed to the individual services and their associated supporting functions and/or processes. While developing a framework designed to assist with understanding AESs, concept maps were developed to illustrate the ecosystem services identified and other complex feedbacks within the system (Pollard et al., 2013). The nature of the feedbacks, however, was not defined in terms of causality, but these concept maps still remain an integral step towards developing more detailed CLDs.

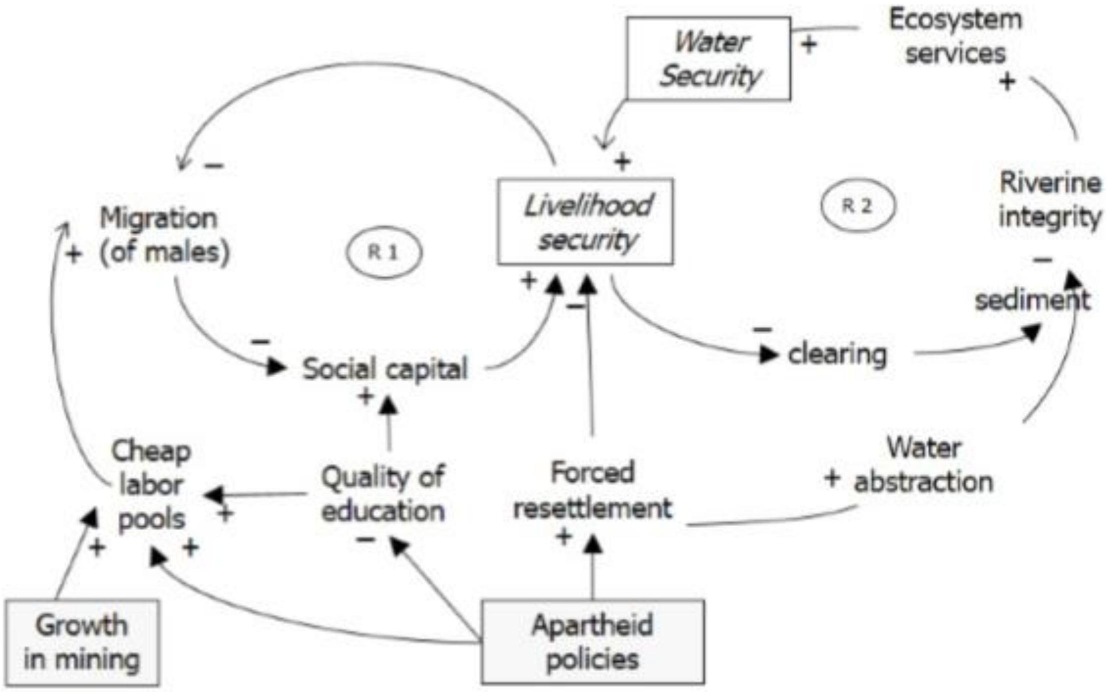

Figure 16: CLD illustrating water security in the Sand River Catchment (Pollard et al., 2014)

Considering the discussion above, it is clear that CLDs could potentially be used to facilitate an alternative VCA for ecosystem services, which will explore the growth path far beyond the subsistence level. Using CLDs will allow for the identification of intermediate and final goods, and services to support the valuation process and improve system understanding.

5.3 Benefits and Limitations of CLDs

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 examining whole systems including all underlying interacting components (Kirkwood, 2013; Sterman, 2000). CLDs facilitate this shift by using 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 (to speak in 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 & 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 because the consequences of ignoring such feedback could be devastating for either system. Table 3 summarises the main strengths and weaknesses of CLDs for system dynamics; however, 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 as verified research in this regard is almost non-existent (Forrester, 1975; Lane, 2008; Morecroft & Sterman, 1994; Richardson, 1996). Lane (2008: 11-14) and Schaffernicht (2010: 665) describe the advantages and disadvantages of CLDs listed in Table 3 in more detail.

Table 3: Advantages and limitations of CLDs (adapted from Lane (2008))

Advantages	Limitations
Focus is on feedback Interest in decision points and performance measures	Loss of distinction 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 an 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 (Lane, 2008; Larkin & 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 several different applications as they are capable of simultaneously and transparently including social, environmental and economic elements (Costanza & Ruth, 1998; Evans, 2004; Kirkwood, 2013; Richardson, 1994).

Hitchins (2005) describes an advantage of using CLDs as forcing the user to tie 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, programme or policy (Evans, 2004; Morecroft, 1982). 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 that can provide essential information to managers and decision makers. It is evident that CLDs can help to understand complex process and system dynamics better, which is vital because system-modelling has grown considerably in many different spheres of influence (Ford, 1999; Senge & Sterman, 1991).

However, CLDs are limited in many aspects and should be thoroughly understood before completely relying on them to understand complex, nonlinear system processes. One of the primary issues with CLDs is that they fail to make a distinction 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). Lane (2008), Schaffernicht (2010) and Sterman (2000) 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, many prefer the ‘truncated’ definition: “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 literature for its 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 generally 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 how these influences work (Kirkwood, 2013; Schaffernicht, 2010). Thus, the relationship between event and behaviour is regularly counterintuitive. If the definition of polarity facilitates faulty conclusions or is too complex to be used properly, 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 general but fundamental limitations of CLDs, which coincide with those described by Lane (2008) (Table 3). CLDs are a poor monitor of behaviour if the system contains more than a few causal loops. Devoid of simulation one must rely on intuition to infer the behaviour implied by a structure of the CLD at hand (Forrester, 1994; Lane, 2008; Richmond, 1994).

Furthermore, the *ceteris paribus* condition (explained earlier) applied to CLDs and most qualitative modelling approaches can be considered a shortfall as there are always more variables that 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 cannot accurately represent cause and effect over time (despite the inclusion of delay marks), which causes one to implicitly assume that the effect of an intervention on a particular control variable will be realised immediately, which 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 each model (Ford, 1999; Sterman, 2000). Issues surrounding the spatial dimension of CLDs can be accounted for to a certain degree by incorporating GIS[1] maps that relate to the elements of the system. CLDs can 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 the CLD approach is more applicable to certain applications than others. This serves as further motivation to determine the potential of CLDs to evaluate the value chains of AESs.

6 BAVIAANSKLOOF

6.1 Rationale for Study Area Selection

Baviaanskloof was selected as an appropriate study area for testing the ability of CLDs to assess the value chains of AESs for several reasons. Firstly, there is already a significant amount of scientific research and data available directly related to ecosystem services (Boshoff, 2005; Jansen, 2008; Mander et al., 2010). The large amount of environmental, ecological and economic research focusing on Baviaanskloof means that there are several experts who have first-hand experience in the study area. This is beneficial for hosting specialised workshops as there is a large base of professionals to reach out to. There are also several different user groups, primarily irrigated agriculture, livestock and game farming, conservation and recreation/tourism who compete for these AESs at different levels (Boshoff et al., 2000; Illgner, 2004; Nel et al., 2006). Thus, trade-offs between different ecosystem services for diverse user groups can be incorporated into the VCA.

[1] Geographical information system

Nel et al. (2006) note how Baviaanskloof is a priority biodiversity conservation area for rivers. Additionally, Baviaanskloof is considered to be part of the Kougaberg strategic water source area (SWSA), which includes areas that supply a disproportionate amount of mean annual run-off to a geographical region of interest (Nel et al., 201~~3~~1; 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., 201~~3~~1). Research aimed at developing recommendations to improve aquatic ecosystem functioning will thus be best suited to South Africa’s SWSAs.

The climatic and topographical variations (explained in Section 6.2) in the Baviaanskloof River ecosystems facilitate a large number of different ecosystem services, which are compounded by the fact that most of the land cover is natural veld (Skowno, 2008). The extent of the study area is large enough to consider a diverse spread of AESs while still conforming to the strict definitions outlined above. The ecological management class assigned to 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 nature of the approach being applied in this project and because it has not been attempted before, considering an area that is ‘largely natural’ (minimal anthropogenic disturbances) simplifies the process of identifying appropriate and relevant AESs and describing their complex interactions.

6.2 Physical Attributes of the Study Area

6.2.1 Location and topography

Figure 17 illustrates the location of Baviaanskloof (approximate coordinates range from 23°35’E to 24°25’E and 33°30’S to 33°45’S) within South Africa as well as the quaternary catchments (L81 A–D) that physically delineate the study area. This area is known as the ‘true’ Baviaanskloof as these are the catchments through which the Baviaanskloof River runs despite the Baviaanskloof Mega Reserve (BMR) extending into several surrounding catchment areas as well (i.e. Kouga and Gamtoos) (Boshoff, 2005; DWAF, 2002; Jansen, 2008).

The Cape Fold Mountains account for the rugged terrain that features over most of the study area, apart from the valley floors that are generally flat floodplains typical of highly eroded valleys (DWAF, 2003; Jansen, 2008). The mountains are generally very steep; only one-third of the slopes exhibit a gradient less than 30% (Illgner & Haigh, 2003). The altitude of the mountains north and south of the Baviaanskloof River reaches heights of up to 1758 metres above sea level (masl). The valley floor elevation ranges from approximately 700 masl to 300 masl (Jansen, 2008; Mander et al., 2010). South of the river is the Kouga mountain range; to the north is the Baviaanskloof mountain range. 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) is at the confluence with the Kouga River (Illgner & Haigh, 2003; Jansen, 2008).

6.2.2 Rainfall and climate

Baviaanskloof is situated in a bimodal rainfall zone with spring and autumn maxima, with an annual average of approximately 300 mm and a large interannual variability (ranging from less than 100 mm to more than 700 mm) (Mander et al., 2010). The rain is primarily convective and/or orographic in nature with more than two-thirds falling in the summer months (Jansen, 2008).

Similarly, average temperatures exhibit high interannual variability ranging anywhere from 32ºC to −3ºC (Boshoff, 2005; Mander et al., 2010). Frost occurs frequently during winter months, as well as snow on many of the mountain tops. During winter months, the prevailing wind direction is northwest; during summer, it is southeast (Jansen, 2008).

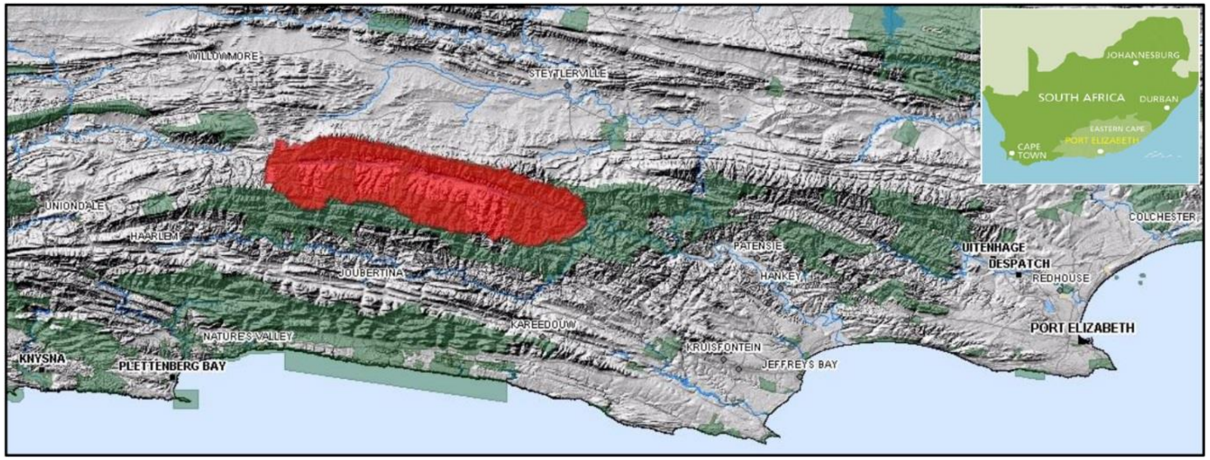

Figure 17: The Baviaanskloof

6.2.3 Geology

Quartzites and sandstones of the Table Mountain Group (TMG) make up the dominant geological formation of Baviaanskloof with small amounts of shale that originate from the Bokkeveld Group. Both formations have been subject to intensive folding (Illgner & Haigh, 2003; Rust & Illenberger, 1989; Welman & 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 & Haigh, 2003; Jansen, 2008; Welman & Barnard, 2007). The arenaceous TMG rocks make up the vast majority of the Kouga and Baviaanskloof mountains. The valley floors consist mainly of shales from the Bokkeveld Group (Welman & Barnard, 2007). Stratified sandy alluvial soils originating from the fine-gained sedimentary nature of the Bokkeveld Group are dominant in the valley, exhibiting high porosity and drainage (DWAF, 2003; Welman & Barnard, 2007).

Welman and Barnard (2007) describe the hydrogeology of Baviaanskloof primarily by 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, which are relatively high rainfall areas that support groundwater recharge (Welman & Barnard, 2007; Yong~~ Wu~~, 2005). The alluvial aquifer is considered 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 & Barnard, 2007). The aquifer makes direct contact with the Baviaanskloof River and is recharged via run-off from mountain slopes and rainfall in the valley. This allows the aquifer to the maintain base flow of the river during dry periods (Jansen, 2008; Welman & Barnard, 2007). Much of the surrounding agriculture uses water from the alluvial aquifer for irrigation purposes. There is no evidence of whether the two aquifer systems are directly connected or not (Jansen, 2008).

6.2.4 Surface water and surrounding catchments

Figure 18 illustrates the delineation of and interrelations between the Gamtoos, Groot, Kouga and Baviaanskloof catchments; the hydrology of each catchment is directly influenced by those adjacent. The Kouga and Baviaanskloof Rivers flow into their respective catchments and meet at the Kouga Dam, from where the water is released into the Gamtoos catchment. Water flows from the Groot catchment into the Gamtoos catchment. Small amounts of water flow peripherally into the Baviaanskloof catchment as well. 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).

The Baviaanskloof River (Figure 18) is not always visible throughout the year as some river sections continue as ephemeral interflow through permeable sections of alluvial deposits (Jansen, 2008). Baviaanskloof is a perennial river with a number of non-perennial tributaries (Figure 17 and Figure 18) 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).

6.2.5 Land use

The largest portion of land (approximately 65%) in Baviaanskloof is used as a nature area (BMR), which is used for watershed and biodiversity management. The rest of the land is used for agriculture, game farming, settlements and tourism (Boshoff, 2005; DWAF, 2002; Kirkman, 2006). Farm sizes vary significantly and most agricultural land is used for livestock cultivation (subsistence based), primarily goats and sheep. A very small portion of land is used for irrigated agriculture. Most who irrigate rely on natural springs, boreholes dug into the valley, and small farm dams (Boshoff, 2005; Boshoff, et al 200~~8~~0; Jansen, 2008; Kirkman, 2006). Ecotourism has increased recently as much of the agriculture in the area is decreasing due to the attractiveness and remoteness of the area (Boshoff, 2005; DWAF, 2002).

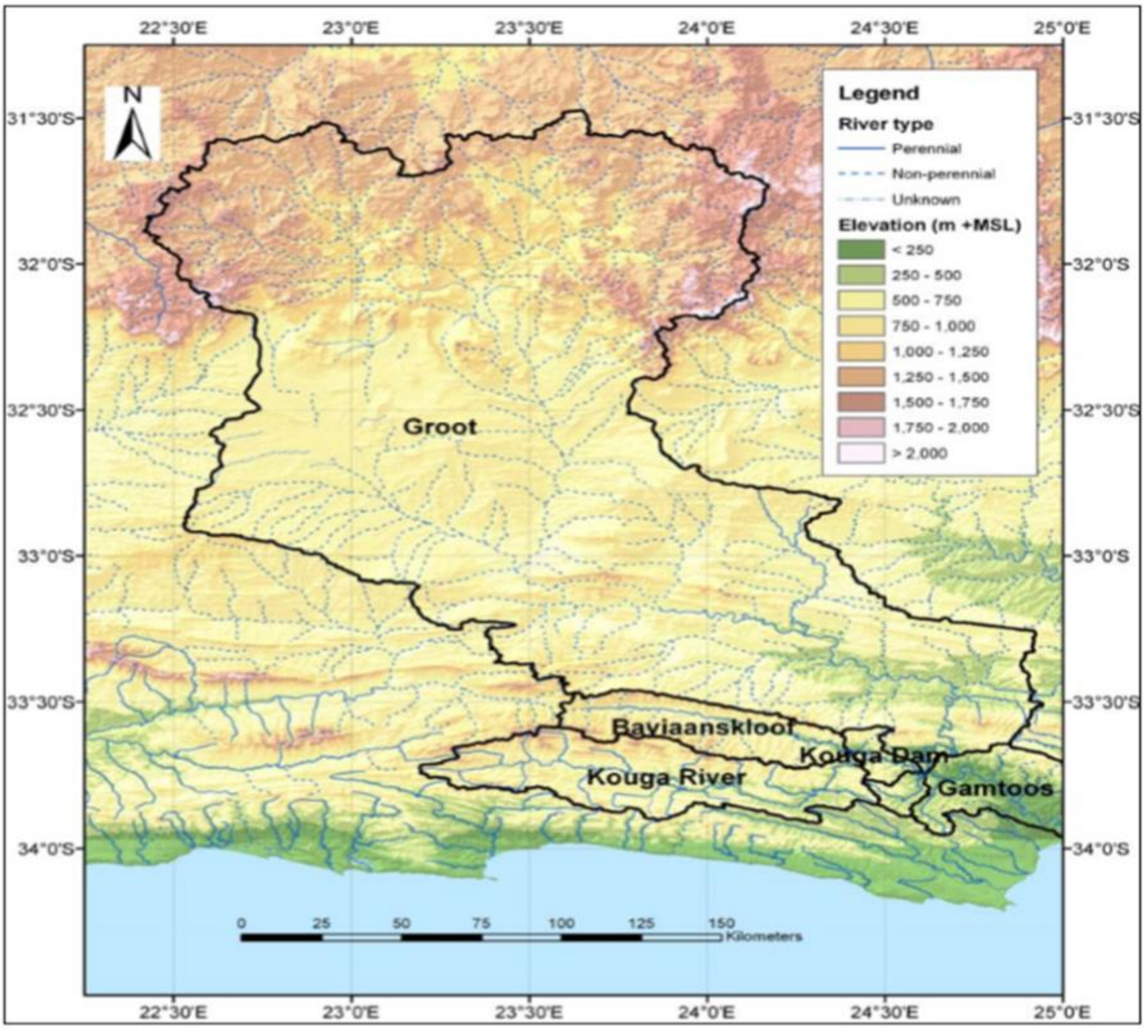

Figure 18: Catchments surrounding Baviaanskloof (adapted from Jansen (2008))

6.2.6 Vegetation

Baviaanskloof falls within the eastern end of the Cape Floristic Region and is thus part of the smallest and most distinctive of the six plant kingdoms (Boshoff, 2005; DWAF, 2003). The Grassland, Subtropical Thicket, Nama-Karoo and Fynbos biomes occur marginally in the Baviaanskloof, while seven of South Africa's eight biomes are considered to occur in the broader Baviaanskloof (Boshoff, 2005; Kirkman, 2006). The general vegetation types are dominated by fynbos and subtropical thicket components (Boshoff, 2005). Mander et al. (2010) describe the complex vegetation community, structure and composition of Baviaanskloof as being dominated by fynbos and biome-transitional vegetation types.

7 A CLD FOR AESs IN THE STUDY AREA

Some working definitions were required to identify AESs in the study area. After consulting literature, it was decided to adopt the following definitions from Johnston and Russell (2011):

- **Final ecosystem services:** biophysical outcomes that directly enhance the welfare of at least one human beneficiary.
- **Intermediate ecosystem services:** biophysical conditions, functions or processes that only benefit humans indirectly through effects on other ecosystem services (biophysical outcomes) first.

- **AESs:** The aspects of aquatic ecosystems used (actively or passively) to provide benefits that contribute towards the welfare of one or more human beneficiary groups.
- **Aquatic –** relating to freshwater.

To allow the construction of the CLD later, the working definitions need to be supported by decision rules to distinguish between final and intermediate ecosystem services, which present the basis for causal relationships and direction of flow within the CLD. These two properties later form the basis of the AESs value chains. Again, we employed Johnston and Russell (2011) for these definitions:

- 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.
- 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.
- For endpoint *h* to serve as a final ecosystem service for beneficiary *j*, the beneficiary must be **willing to pay** for increases in *h*, holding all other ecosystem outputs and conditions $I \neq h$ constant.

An expert workshop was held where the above-mentioned decision rules were applied to identify AESs in the study area in accordance with the working definitions. The following individuals were consulted:

- 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 Living Lands Research Assistant.

Other specialists consulted:

- Prof. Jane Turpie: University of Cape Town.
- Dr Dieter van den Broeck: Director of Living Lands.
- Dr Patric O'Farrell: Principal Scientist, CSIR.
- Dr David le Maitre: Principal Scientist, CSIR.

Experts were asked to identify beneficiary groups and associated benefits derived from final AESs using the methodology described by Johnston and Russell (2011). Final AESs realising each benefit were then identified. The processes (intermediate services) required for each AESs were identified and described via an iterative backtracking process which continued until primary processes were identified (i.e. processes that affect several AESs).

Table 4 gives the beneficiaries, benefits and final AESs for the study area.

Table 4: Beneficiaries, benefits and final AESs as identified for the study area

	Beneficiary Group	Benefit (Improves Welfare)	Final Ecosystem Service
1	Farmers	Water for Agricultural Use	Water Provision
2	Residents	Protection from flood damage	Flood Attenuation
3	Conservationists	Aquatic Biodiversity	Aquatic Ecosystem Health
4	Tourists	Aesthetic/Spiritual Benefit	Aquatic Ecosystem Health
5	Domestic Users	Domestic Water for Consumptive Use	Water Provision

This information was used as the basis for constructing the CLD. Vensim software was used to graphically illustrate the identified final and intermediate ecosystem services associated for each service. A discussion then followed to identify and describe the direct and indirect interactions and relationships between each service (final and intermediate). These causal relationships were systematically added to the diagram (Figure 19).

On the CLD, the green circles indicate the final AESs, the white boxes indicate intermediate ecosystem services, and the yellow boxes indicate the anthropogenic components. Two lines on an arrow indicate a delay in the causality between the two components (e.g. rainfall and vegetation).

Due to the CLD being fairly large already, many of the terms on the diagram could have multiple meanings, hence the need to describe each component accurately to understand exactly how they are linked to one another. The purpose was not to create a comprehensive list of AESs and the intermediate components that affect them, but rather to develop a relatively realistic representation of a complex system and to test the CLD as a tool for VCA. Thus, the aim is to identify the most important intermediate processes and functions that affect the identified final ecosystem services and determine how they impact or are impacted by each other on the diagram (Table 5).

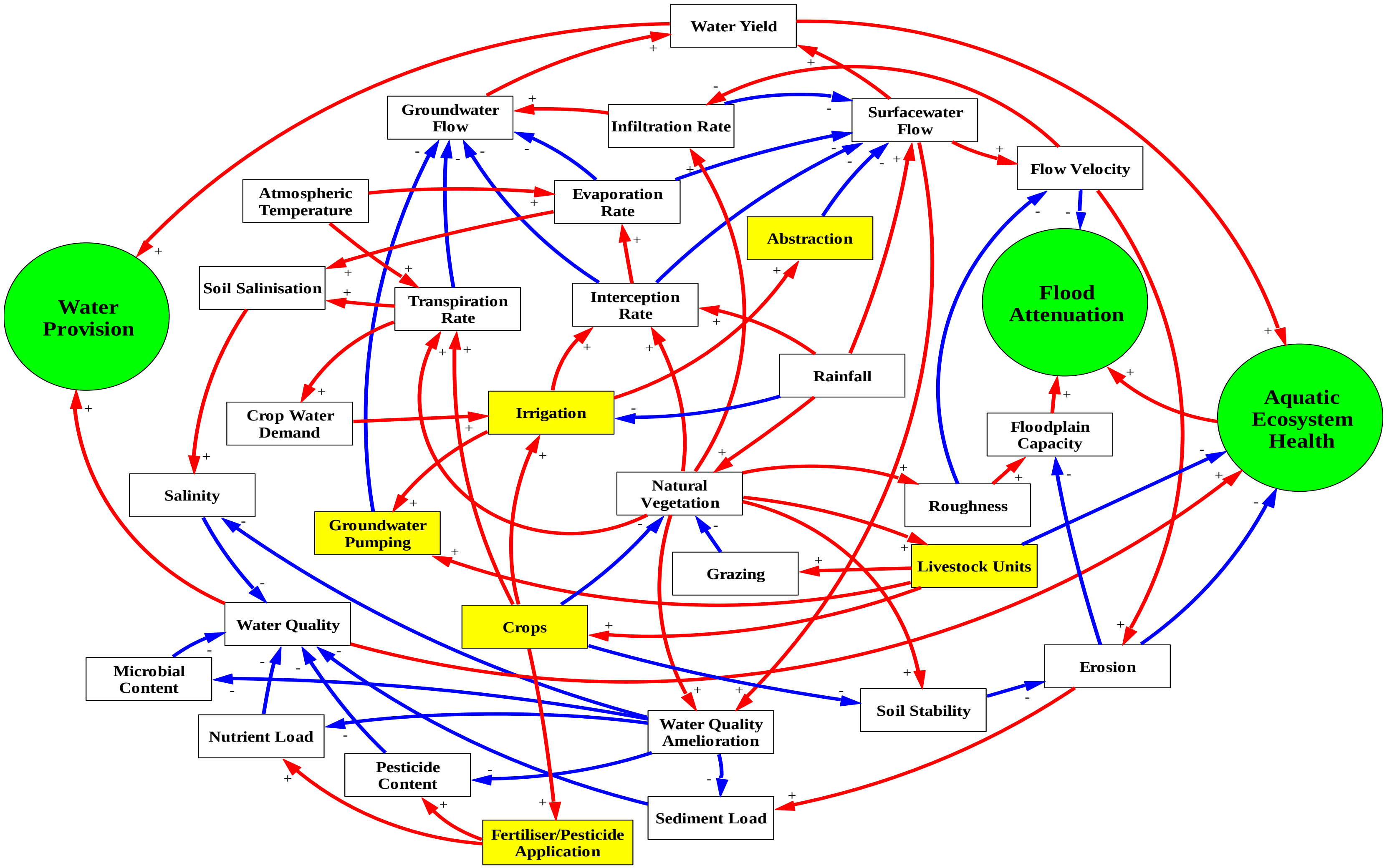

Figure 19: CLD for AESs in the study area

Table 5: Description and unit of measurement for intermediate and final ecosystem services

Intermediate Aquatic Ecosystem Service	Description	General Unit of Measurement
Groundwater Flow	Water present beneath the Earth's surface over time.	Volume/Time
Surface water Flow	Water present on the Earth's surface over time, which includes channel and overland flow.	Volume/Time
Water Yield	Total water runout from the two catchments, which includes all ground and surface water.	Volume/Time
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)
Evaporation Rate	Process of water changing state from liquid to water vapour due to increased temperature and/or decreased pressure.	Volume/Time
Transpiration Rate	Process of plants converting water into water vapour, which is released into the atmosphere.	Volume/Time
Infiltration Rate	Process of water on the ground surface entering the soil.	Volume/Time
Crop Water Demand	Amount of water required for a cultivated crop to meet the water loss through evapotranspiration.	Volume
Atmospheric Temperature	Measure of the temperature at the surface of the Earth.	Degrees Celsius
Interception Rate	The process of leaves and branches of plants preventing precipitation from reaching the ground (soil).	Volume/Time
Roughness	Floodplain/channel shape and material texture that affects flow velocity through friction.	Manning's Roughness Coefficient (Limerinos, 1970; Li & Zhang, 2001)
Floodplain Capacity	Space available in the floodplain and surrounding area capable of retaining water.	Volume
Flow Velocity	The rate at which water flows in a channel or over the surface of the Earth.	Velocity
Natural Vegetation	Amount of natural vegetation (biomass) present.	Mass
Soil Salinization	Biophysical process of increasing soil salt content.	Mass/Volume
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)
Rainfall	The quantity of rain falling within a given area over a specific time period.	Volume
Erosion	Action of exogenic processes that remove soil and rock from the Earth's surface.	Mass/Time

Intermediate Aquatic Ecosystem Service	Description	General Unit of Measurement
Salinity	Measure of all the salts and substances dissolved in water.	Mass/Volume
Microbial Content	Pathogenic microorganisms in surface and groundwater.	Mass/Volume
Nutrient Load	Phosphates and nitrates in surface and groundwater.	Mass/Volume
Pesticide Content	Pesticides and herbicides in surface and groundwater.	Mass/Volume
Sediment Load	Solid matter carried in suspension in surface and groundwater.	Mass/Volume

Anthropogenic Components	Description	Unit of Measurement
Livestock Units	Number of livestock units in a specific area over a specific period of time.	Livestock Units/ Area
Pumping	Artificial transport of groundwater to the surface for irrigation or livestock use.	Volume
Crops	Cultivated plants grown for commercial or subsistence purposes.	Area
Irrigation	Artificial application of water to land/or soil.	Volume
Abstraction	Process of extracting surface water for anthropogenic use.	Volume
Effluent Disposal	Human waste disposed directly into a watercourse.	Volume
Fertiliser/Pesticide Application	Fertiliser and/or pesticide applied to cultivated crops.	Volume

Final Aquatic Ecosystem Service	Description
Water Provision	The volume of water provided by the ecosystem suitable for consumptive use. This is determined by the yield and quality of the water.
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.
Flood Attenuation	The ability of an ecosystem to retain water in situ and release it over time, reducing flood occurrences and damage.

The calculation of forward multipliers of selected AESs requires longitudinal data on market transactions of established markets. None of these exist for AESs and the multipliers cannot therefore be quantified in the traditional sense, but will instead be described within the CLD (the market-making process for AES are discussed in Section 12).

8 SCENARIO ANALYSES

Thus far we have presented relevant literature reviews, the methodological approach for constructing the CLD for AESs in the study area, and the CLD itself. The scenario analysis for presenting the ripple effects of external shocks on the CLD and associated recommendations to improve (increase) the resilience of AESs delivery will be discussed next.

It was mentioned that a traditional approach towards multiplier analysis does not apply to the context of this study, mainly because of the public nature of the service and the associated absence of formal market data (input cost structure and market-clearing prices). A broader interpretation of 'multiplier effects' was required. Consequently, we assessed the ripple effects of external 'shocks' on these services in its broadest sense by using a CLD specifically developed for the project. The CLD presented in the previous section was used to illustrate the positive and negative knock-on (ripple) effects due to changes in the value chains of the AESs. Changes were simulated by means of selected scenarios, which represent different external 'shocks' to the current state of the system. The scenarios were chosen carefully and developed with the real-life context of the study area in mind. The consequent linear causal pathways of the shock(s) are presented in the CLD. Although the causal pathway is not quantified or fully inclusive, it is considered adequate for the purposes of recommending improvements of aquatic service delivery and its markets as per Section 11. The main outcome of the CLD development phase of the study (Section 7) has demonstrated the various processes and functions that contribute towards the provision of the three final AESs, namely, water provision, flood attenuation and aquatic ecosystem health (see Figure 20). A descriptive summary of the final and intermediate AESs along with the environmental variables and anthropogenic variables affecting these are presented in Appendix 1.

The CLD demonstrates how the complex array of intermediate AESs affect the provision of the final AESs in a multidimensional snapshot. Each causal linkage (arrow) qualitatively indicates the relationship between the two variables it connects. These relationships effectively demonstrate the forward linkages in the value chain components of the final AESs.

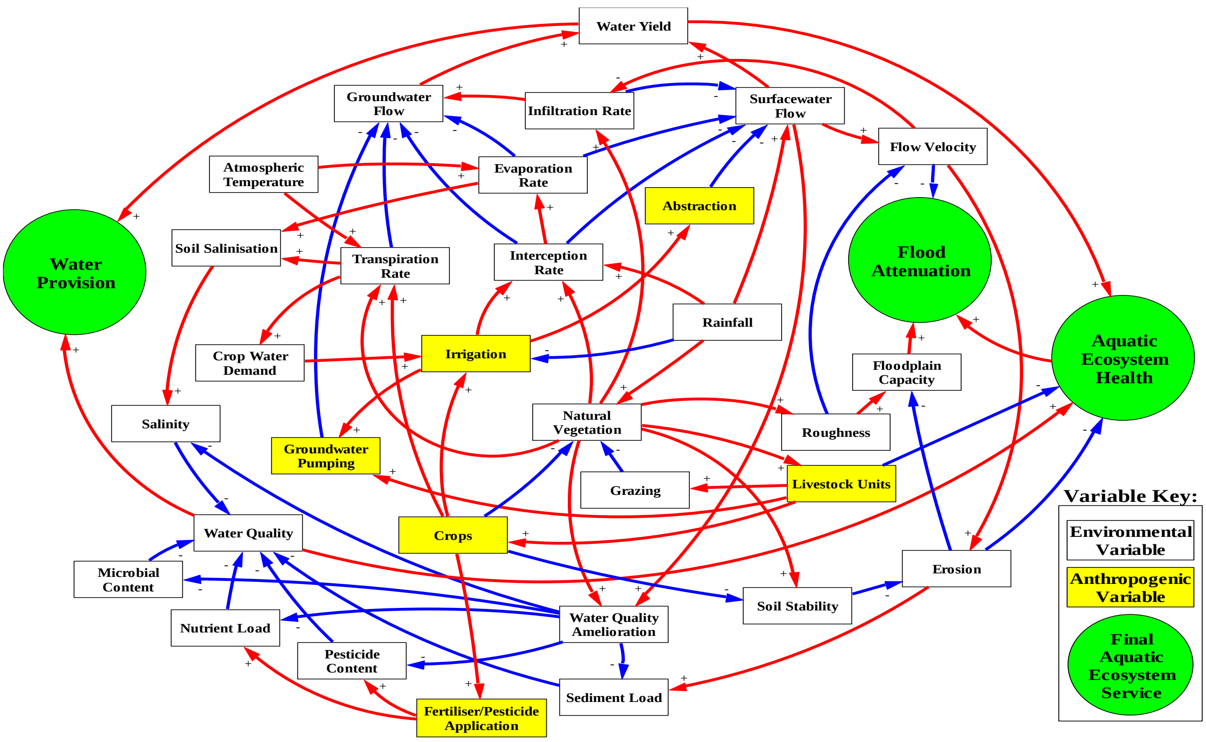

Figure 20: CLD illustrating select AESs, processes and functions in the Baviaanskloof catchment

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 could fit into the different CICES categories. In order to address the aim of the project, 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. Also, there is 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 in 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 in formal markets.

No positive or negative feedback loops were identified due to the complex and dynamic nature of the system and the tiered structure of the intermediate ecosystem services. 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 VCA 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 was determined by analysing the impact of different scenarios on the system by identifying linear causal pathways (individual value chains) that demonstrate how various disturbances influence the provision of the final ecosystem service at hand. Thus, allowing potential challenges and/or opportunities to be identified in these individual value chains, regardless of how integrated the service is in formal existing markets. Simultaneously, the ripple effects of the various disturbances to the system are illustrated in terms of their impact on the provision of the final AESs. However, although the ripple effects in the system are illustrated, it was not possible to present 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 the amount of 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, which is to determine the impact of a change on a system in isolation. Significant more work will be required to enable such modelling since it requires the system to be become dynamic, i.e. quantitative system dynamics modelling would be required to model these impacts, which was considered unnecessary for serving the purpose of the study.

The scenario analyses were designed to illustrate how different scenarios lead to positive or negative impacts on the system as a whole and how this affects the provision of the final AESs in a multi-dimensional way. From here, the VCAs took a linear form by presenting the causal impact in a linear casual pathway.

Four different scenarios in the Baviaanskloof were analysed, namely, fire (Section 8.1), drought (Section 8.2), land use change (Section 8.3), and catchment restoration (Section 8.4). Fire and drought are naturally occurring phenomena while the remaining two scenarios are anthropogenically induced. These four scenarios were chosen as they are realistic scenarios within Baviaanskloof and will thus have representative and practical management and decision-making outcomes. All four scenarios occur over 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 time scale. The varied temporal scale and differences in the manner in which they occur were primary reasons for choosing these scenarios.

It is critical to describe each scenario accurately and in detail when conducting the analysis. Although the scenarios vary from one another, the same scenario could have a multitude of impacts on the system in different circumstances. For example, a fire has the potential to directly impact natural vegetation, crops and soil stability.

The scenario analysis begins with the disturbance to the system (e.g. fire). It then logically follows the impact of the disturbance through the various linkages until the nature of the impact on the final AES can be determined. The physical outcome of such an analysis is a CLD that illustrates the effect of the disturbance on the system using highlighted arrows. The immediate, short- and long-term impacts are shown via different arrow colours in an attempt to compensate for the diagrams' limited ability to illustrate differences in time scales. Specifically, 'short term' refers to impacts that occur within days or weeks of the disturbance, while 'long term' considers months to years. The four different scenarios and their outcomes are discussed and illustrated in the sections that follow.

8.1 Scenario: 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 phenomenon in Baviaanskloof and plays an important role in veld management in the area (Boshoff et al., 2000; 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 (Booysen & Tainton, 2012; Glenday, pers. comm., 2015; Kruger, pers. comm., 2015). This scenario in particular replicates 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 occur from this event over three aforementioned time scales (immediate, short-term and medium/long term) as exemplified in Figure 21, which demonstrates how the disturbance variable impacts all three of the final AESs.

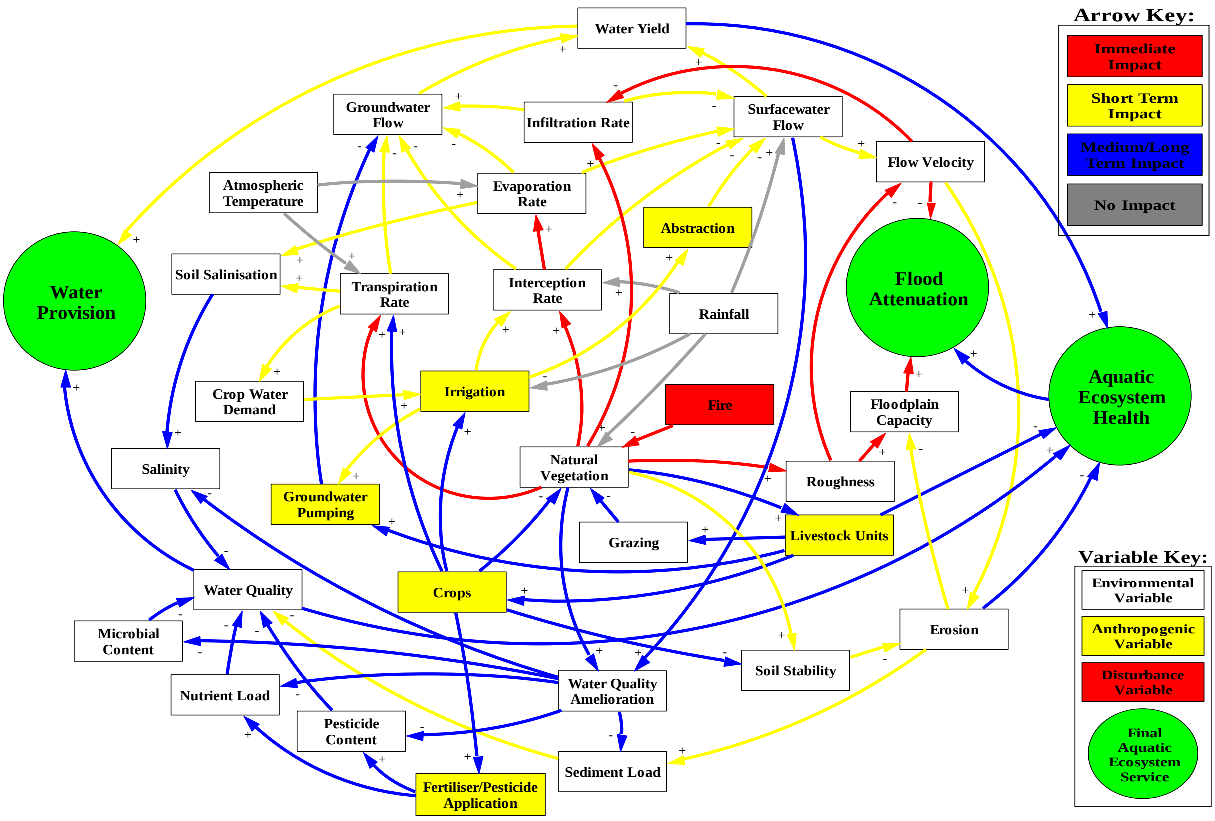

Figure 21: Fire scenario analysis CLD for AESs in the Baviaanskloof catchment

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. 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 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 consider any knock-on effects such as the regrowth of vegetation over time, which would counteract these negative impacts.

8.2 Scenario: Drought

Weather patterns around the world are increasingly being influenced by climate change. As a result, severe weather events such as droughts and floods are becoming more prevalent (Houghton, 2004; IPCC, 2013). Although most farmers in 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). Drought is a relatively common occurrence in Baviaanskloof. There is often less than 100 mm rainfall per year (Kruger, pers. comm., 2015; Mander et al., 2010). The drought scenario is an illustration of a long-term occurrence that attempts to simulate a decrease in the amount of rainfall over a long time scale. Figure 22 illustrates the impact of drought on the system highlighting the different time scales of impacts. Despite drought occurring over a long period, the impacts were considered to occur over a range of time scales.

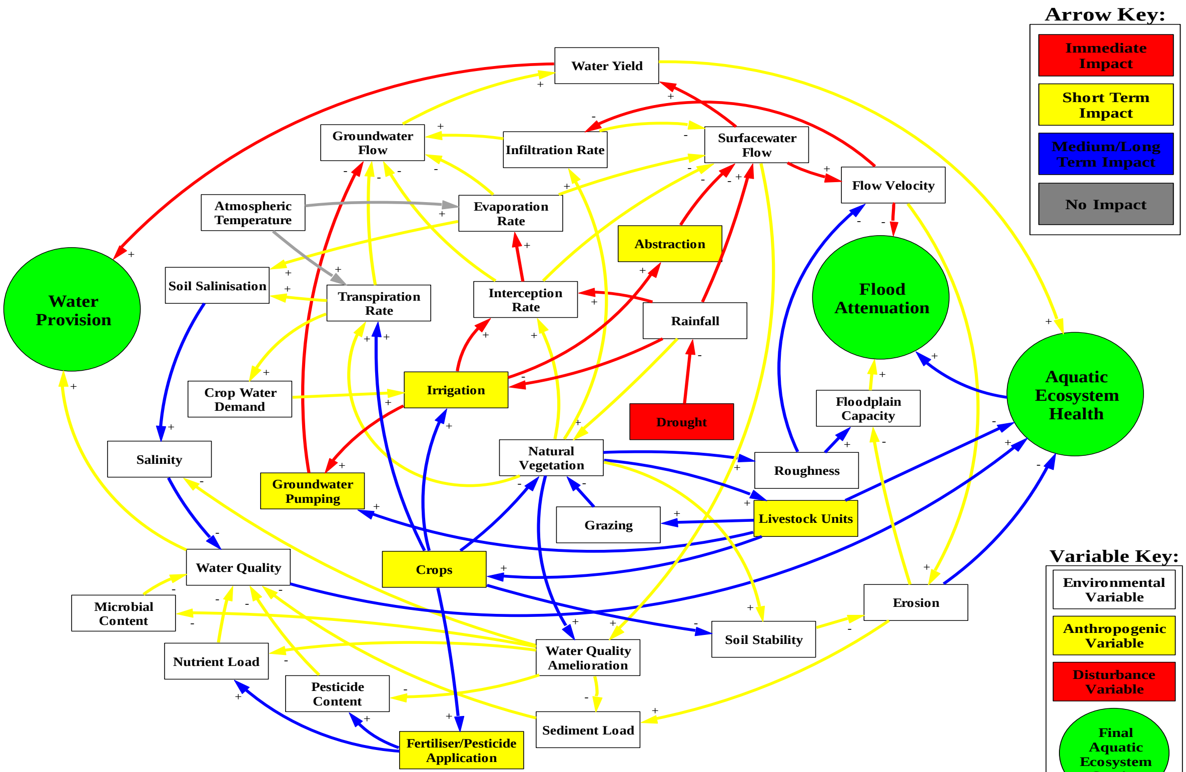

Figure 22: Drought scenario analysis CLD for AESs in the Baviaanskloof catchment

From Figure 22 it is evident that a drought would have a negative impact on water provision within an immediate to short-term time frame. This is logical regarding 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 over the short to medium term. The general aquatic ecosystem health also decreases 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 made regarding the direction in which flood attenuation would be affected due to the qualitative nature of the model.

8.3 Scenario: Land Use Change

The land use change scenario was based on a proposed business plan by two organisations within Baviaanskloof. The aim of the project is to assist local farmers in converting from growing fodder crops for their livestock to growing lavender for essential oils. While the scheme would be run on a profit-sharing basis, all seed and plant material will be supplied by the development corporation at no direct cost to the local farmers. The ultimate aim of the project is to slow the flow of water through the system (promote diffuse flow) and create an alternative income steam for vulnerable farming communities. As lavender has a comparatively lower water demand than lucerne (the main fodder crop), it will reduce water use in the Kloof. However, livestock densities will need to decrease (i.e. an indirect cost for the farmer) as famers will be switching from lucerne to lavender. The project will therefore 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.

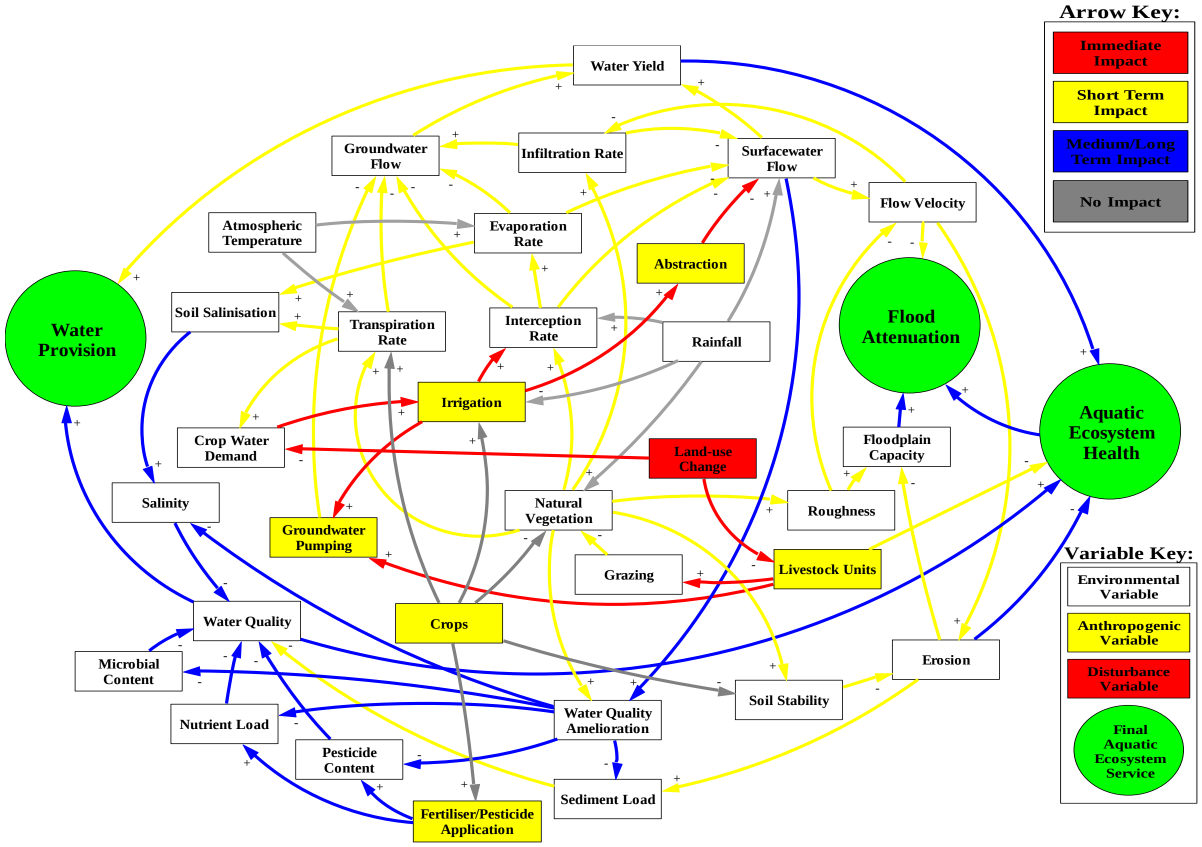

Figure 23: Land use change scenario analysis CLD for AESs in the Baviaanskloof catchment

It is clear from Figure 23 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 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. The ability of the system to attenuate floods will probably decrease in the short to medium term because of increased flow velocity, but will once again improve eventually when the roughness and floodplain capacity have increased. Less livestock and more water available will also increase the general aquatic ecosystem health in the medium to long run. It is evident that the aims and objectives of this project are theoretically supported through the CLD.

8.4 Scenario: Catchment Restoration

The catchment restoration scenario is a demonstration of some of the restoration activities currently being performed in the Baviaanskloof catchment. The Subtropical Restoration Programme and Working for Water (WfW) are the primary proponents of these restoration initiatives (Boshoff, 2005; Mander et al., 2010). These activities include building weirs, stabilising the riverbank with gabions and small balancing dams, restoring alluvial fans, and planting *Portulacaria afra* (*spekboom*) in degraded areas (Illgner & Haigh, 2003; Mander et al., 2010). The ultimate purpose of these activities is to promote diffused flow of water throughout the catchment to retain as much water as possible for as long as possible.

Considering all 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 24 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.

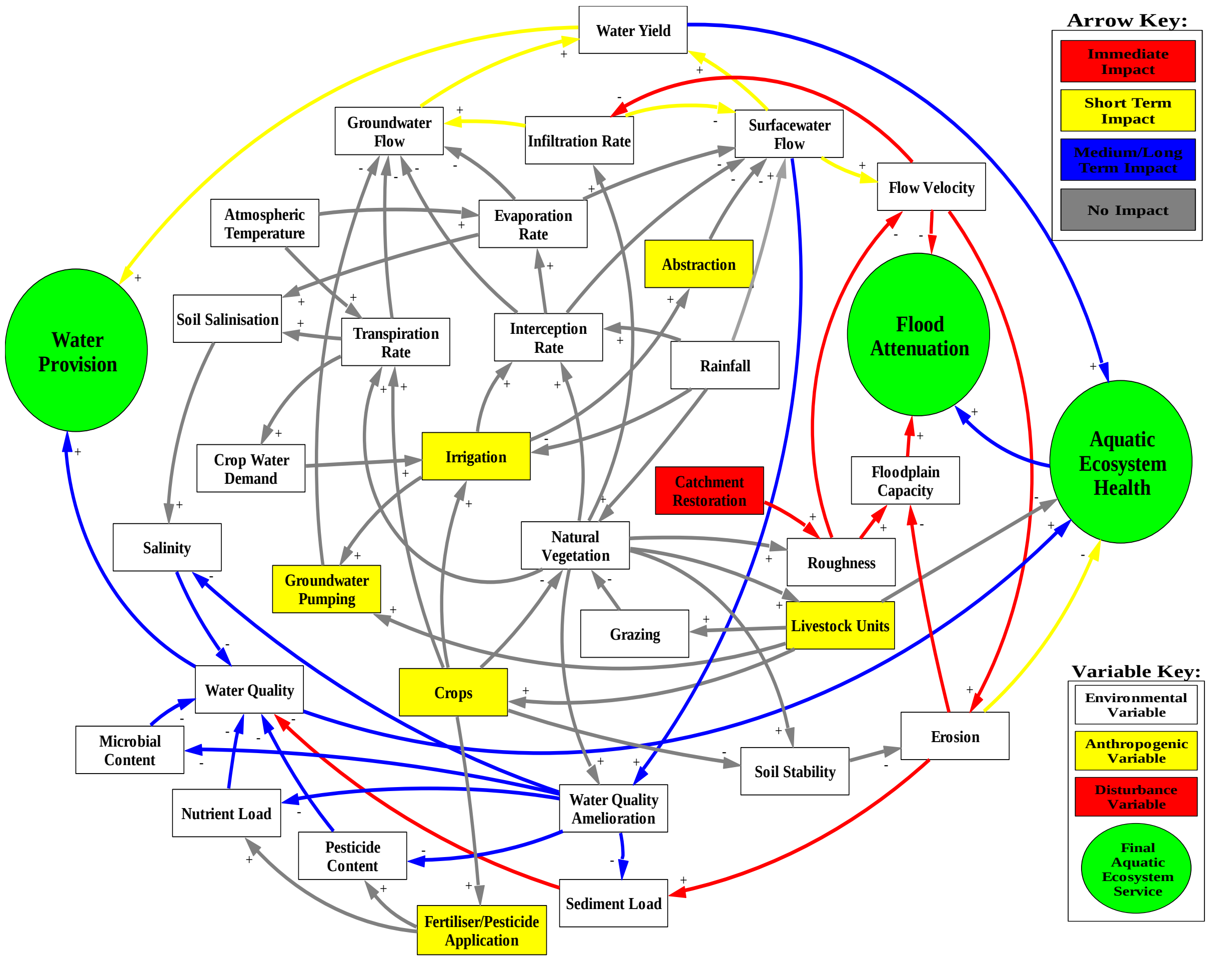

Figure 24: Catchment restoration scenario analysis CLD for AESs in the Baviaanskloof catchment

Regarding 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, the system's ability to attenuate floods will be increased in the immediate term and an immediate decrease in sediment load is met by a medium-term decrease in dilution capacity, which has opposite impacts in water quality; hence one cannot deduce the nature of the impact on water quality over a long period. The impacts on aquatic ecosystem health are once again ambiguous as it should improve in the short to medium term due to decreased erosion and increase further in the long term due to increased water quality.

9 VALUE CHAIN ANALYSES

Further to Section 4, VCA has become an increasingly valuable approach to develop a detailed outlook of the various stages and variables involved in taking a good or service from raw materials, through production 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 & Morris, 2000). The concept of VCA was developed to create a competitive advantage for private sector entities by maximising selected activities in any of the variables in the value chain (IMA, 1996; Kaplinsky & Morris, 2000; Porter, 1985; Schmitz, 2005).

This objective can be compared to improving the output or increasing the provision of final ecosystem services. However, public goods are not constrained by having this bottom line as 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 ecosystem service beneficiaries.

Furthermore, the VCA concept could also be used to illustrate the underlying processes and functions that contribute to providing the ecosystem services that private enterprises rely on, but often take for granted. 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 relies on to function and remain competitive, this would act as an incentive for private and public investment in the sustainable management of these natural systems. The result of this process is to develop extended, more inclusive value chains that formally integrate ecosystem services into established markets for goods and services that depend on their provision.

When attempting to conduct a VCA for complex natural systems that incorporate multiple variables and nonlinearities it is imperative to take a holistic approach. This makes it impossible to perform a traditional, linear type of analysis, as there is no defined starting point for any ecosystem service value chain. 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. Hence the need for scenario analyses to provide a starting point for a VCA. 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 25 illustrates an attempt to create a traditional linear value chain for water provision from the processes and functions in the CLD of Figure 20. The purpose is to illustrate how it is possible in theory to create comprehensive linear and traditional value chains that model complex systems. However, by ‘forcing’ the traditional approach to VCA on an intricate ecosystem, it should be 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 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 ecosystem service. As a result, the chain would become even more convoluted; especially considering how the natural feedback mechanisms (illustrated via the blue interlinking arrows) would make the linear chain loop back on itself. Furthermore, this example only considers the variables presented in Figure 20. 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. Ultimately, it is evident that a traditional linear VCA approach would not be an appropriate management tool capable of effectively representing complex socioecological systems. Hence, using CLDs and scenario analyses to allow for an alternative approach towards VCAs to be conducted for these systems.

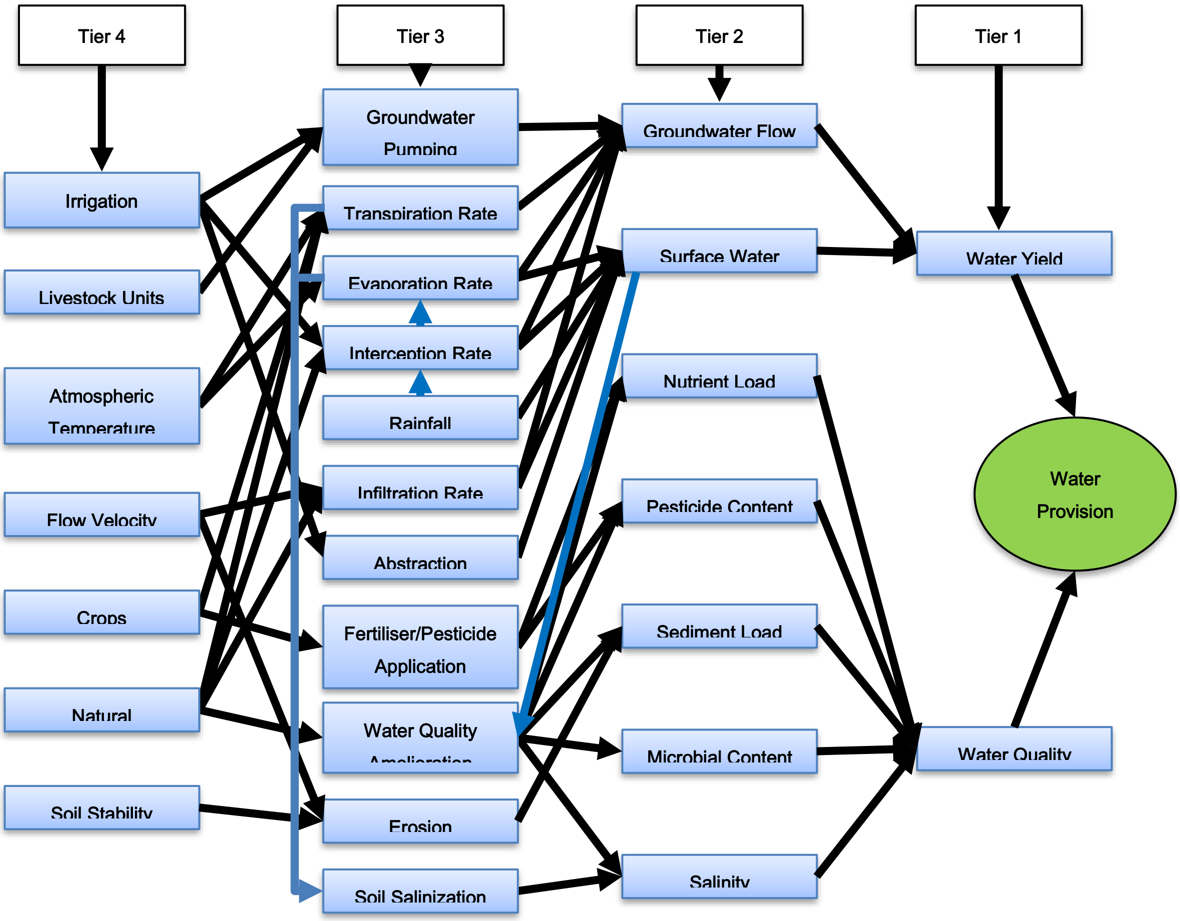

Figure 25: Four tier traditional value chain example for water provision

The alternative VCAs presented here use a demand-side approach (working from the final AES backwards) to identify individual linear 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) to the impact on the final AES. When there are multiple casual pathways affecting the same variable in opposite directions, then no robust conclusions can be drawn regarding the impact on the final ecosystem service. However, informed deductions can be made about the time scale of the impact, and potentially of the magnitude of the different impacts if the catchment dynamics are well understood. Through this process, specific value chains can be identified that are the most relevant to the objective of the management or decision-making processes. This will in turn allow for linear visualisation and understanding of how changes to the system affect the provision of final ecosystem services.

The investigation process that would need to be conducted by management would involve determining the objective of a particular system intervention and/or mitigation initiative (i.e. to reduce the impact of fire on a system's ability to attenuate floods). Then from the scenario analysis diagram, the relevant linear causal pathways (value chains) that flow from fire to flood attenuation would need to be identified. Analyses of these value chains need to be completed to determine the best areas to intervene. The potential leverage points in these value chains can be single or multiple environmental or anthropogenic variables and/or any of the linkages between them.

The example of fire impact mitigation for flood attenuation is discussed in Section 9.2. 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 VCAs examples are illustrated through individual linear causal pathways and discussed for the three final AESs identified in the original CLD below. **The blue arrows illustrate a decrease while the red arrows show an increase in one variable as a result of the change in the previous.**

9.1 Water Provision

Figure 26 graphically presents four individual linear causal pathways that illustrate an example of how each scenario 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 groundwater 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 to narrow it down to linear causal pathways of interest to the objective at hand. This allows logical, coherent and consistent 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, the nett effect of contrasting impacts on the same variable is often an issue of magnitude (i.e. which impact is greater). In these cases, it is not possible to determine whether the increase or the decrease is greater since 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 Water Board 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 26 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 this 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. Private firms interested in maintaining high water yields or mitigating against the impacts of a decreased water yield (i.e. 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.

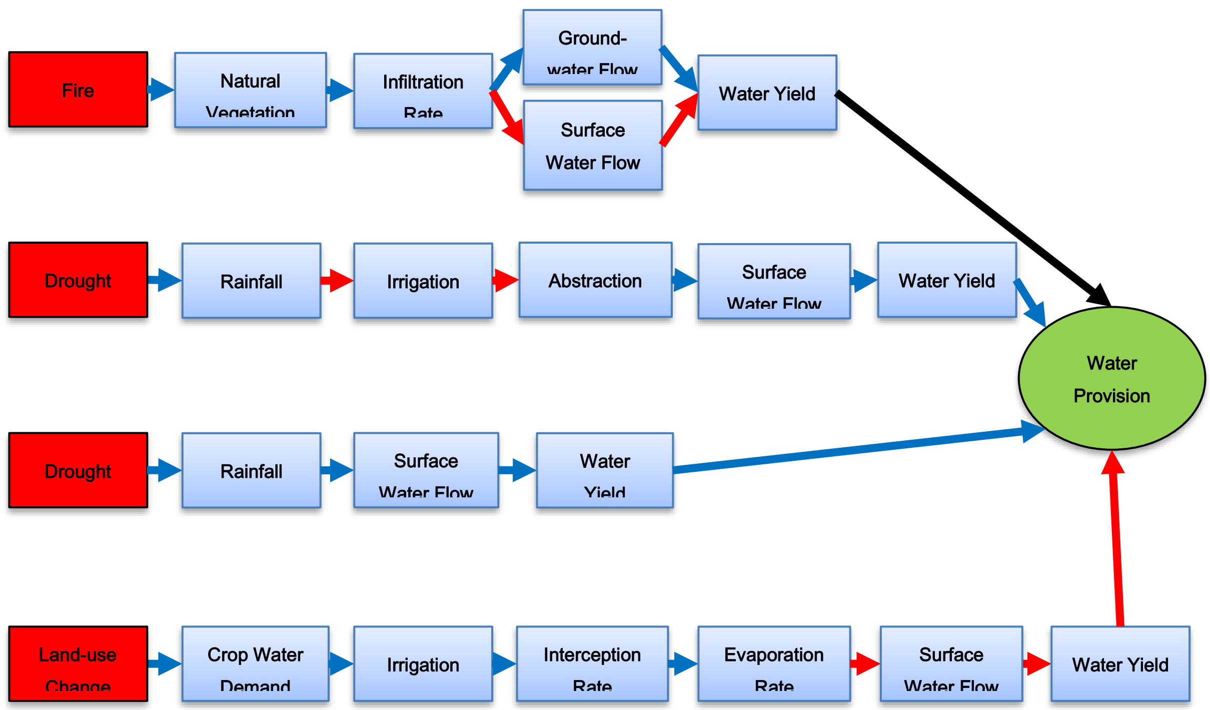

Figure 26: Example of linear causal pathways affecting water provision

This type of analysis is not limited to analysing a single linear causal 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 far easier to repeat. Being able to provide a simplified decision framework to managers and decision makers from the relevant professionals is the most efficient approach.

9.2 Flood Attenuation

Flooding is a relatively common occurrence in Baviaanskloof, making this a realistic outcome for investment companies (Jansen, 2008; Van der Burg, 2008). Figure 27 illustrates three different linear causal pathways that represent the effect of fire and catchment restoration on the system's ability to attenuate floods. Insurance companies are particularly at stake since attenuation could decrease the severity of damage and consequent claims.

Considering the first two linear pathways presented in Figure 27, the loss of natural vegetation because of a fire event indirectly decreases the floodplain capacity and increases flow velocity. Both 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 because of fire, and increase the system roughness and infiltration rate (Booysen & Tainton, 2012). This could be supplemented with a geological survey that identifies the most efficient and effective areas to promote infiltration (i.e. 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 27). If these specific relationships between fire and flood attenuation highlighted in Figure 27 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.

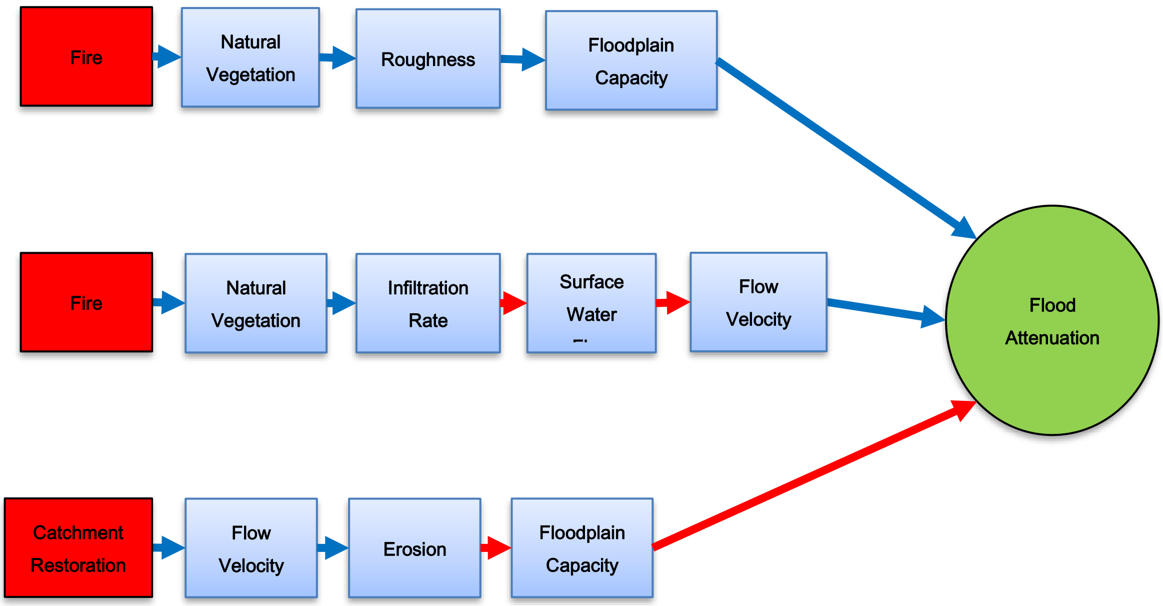

Figure 27: Example of linear causal pathways affecting flood attenuation

Aside from these, other benefits such as improved water retention in the system will benefit local residents.

9.3 Aquatic Ecosystem Health

From the three final AESs under consideration, aquatic ecosystem health is the least integrated in formal markets. Hence, it is difficult to create a scenario that would promote private investment in the restoration and/or conservation of the catchment. It would also be the most difficult final AES to quantify; this would be a necessary requirement to attract private investment (the next section discusses ways to incentivise private sector investment in AESs).

Figure 28 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 28, 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. There are other potential mitigation scenarios 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 thus erosion. From a private investment standpoint, the ecosystem 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 28. There is a multitude of potential mitigation scenarios that 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 20) that has more easily identifiable marketable attributes. This type of final service is often considered as an indirect benefit of management and investment endeavours, but nonetheless has the potential to have significant positive impacts on the system as a whole.

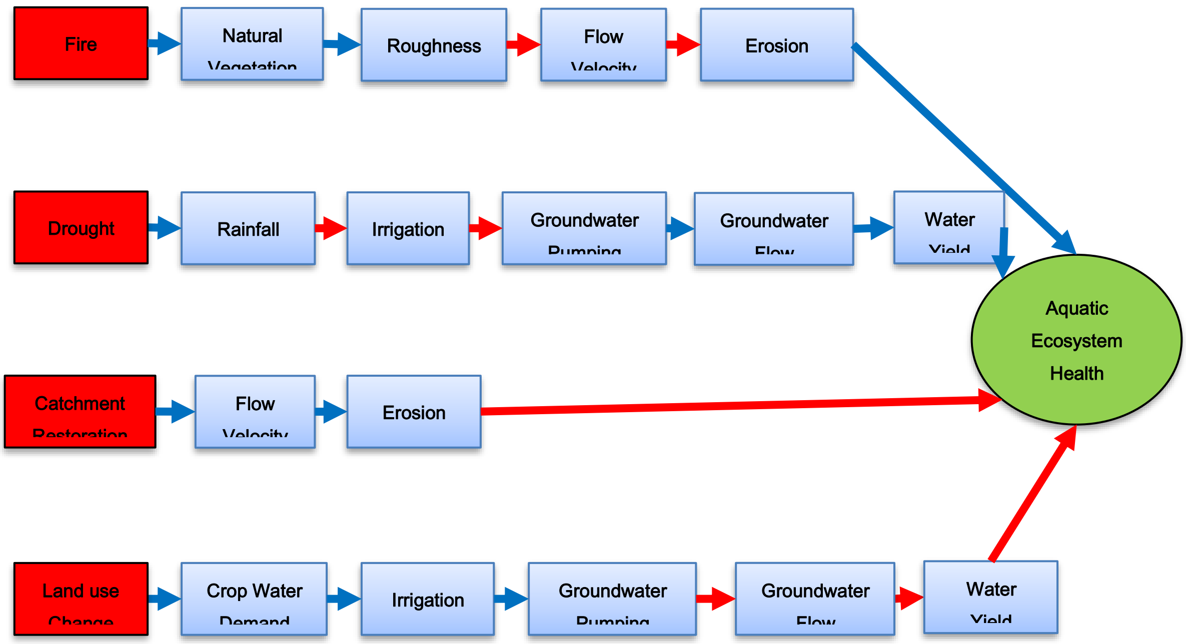

Figure 28: Example of linear causal pathways affecting aquatic ecosystem health

10 STRENGTHS AND LIMITATIONS

It is important to acknowledge and understand the strengths and limitations of the above-mentioned approach to ensure the appropriate use. Firstly, the approach promotes systemic thinking in complex management environments; a need that is widely published crossing many disciplines (Sterman, 2000). It attempts to close the gap between traditional linear economic thought and the complex systems it attempts to model or value. The tool is more inclusive than current environmental management models as it attempts to model environmental as well as anthropogenic components. The model that the approach uses is predictive in nature, thus allowing proactive strategies to be implemented by identifying 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 generating marketed and non-marketed goods and services, makes this approach attractive to the private sector. As previously illustrated, the process has an important role to play to incentivise private investment in the sustainable management of ecosystems, which will not only assist with the integration of these ecosystems into formal markets, but will result in numerous additional benefits associated with healthy functioning ecosystems. The market-making mechanisms for ecosystem services will be assisted through this type of analysis as it outlines linkages between final and intermediate ecosystem services (refer to Section 11).

The approach also 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 ecosystem services as their individual inputs and potential to add value are compared and scrutinised. In theory, this approach can be adapted to a variety of geographical regions in a manner comparable to that of the 'benefit transfer' method, thus making it an extremely versatile method. The method employs a set of dynamic, synergistic tools parametrised towards beneficiaries and end users, which facilitate more informed decision-making regarding complex management challenges.

Although this approach seems to be useful, there are also some limitations that restrict its application in different circumstances. Firstly, it is important to always use the CLDs and the linear causal pathways together to ensure a holistic analysis that considers all the conflicting impacts. The primary limitation associated with a tool of this nature is its inability to account accurately for changes in spatial and temporal scales, despite this being addressed to a certain degree using different colour arrows in the scenario analyses. Additionally, without empirically defining the causal relationships between the different variables, it is not always possible to quantify the magnitude of impacts, which hampers the determination of causality. This is particularly relevant when deducing the direction of the causality where there are two opposite impacts on one variable. Due to the complex and dynamic nature of natural systems, it is challenging 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. The size of the CLD for this study was for the purpose of addressing the aim of this project. The optimal size of the CLD for other purposes may be a more heavily disputed issue.

The nonlinear and ever-changing nature of aquatic systems made it impossible to distinguish between and identify all relevant stock and flow variables identified within the system. Literally every one of the variables change over time. Thus, measurements would need to be taken over a specific time interval to provide meaningful results. Attempting to 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 stock or flow variables in this study. This was the main reason for not initially developing the CLDs from SFDs. Nevertheless, this did not have any impact on the logic flow of the diagram or its ability to represent realistically the system and potential system changes.

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 with the CLD's limited ability to represent changes in time, associated knock-on effects cannot be analysed simultaneously. 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 issue with spatial scale. Technically, variables can continue to be added to eventually include microscopic processes that in some way or another impact the final ecosystem service, 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 most of these issues.

11 TAKING AES TO MARKET

Credit-Suisse et al. (2014) and Lambooy and Levashova (2011) describe the need to move towards public/private investment as well as develop 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.

The remainder of the report focuses on this complex challenge: incentivising private sector investment in aquatic ecosystem service value chains as a specific recommendation to improving aquatic ecosystem service delivery. Several challenges towards increasing private sector investment will be discussed. One of the main challenges lies within the market-making process for AESs itself, which is a major and important obstacle. Special attention will be given to this challenge where the theory of the market-making process will be discussed before moving on to discuss the actual market-making process according to the following steps: basic characterisation; terms conditions and transactional protocol; monitoring systems; piloting; and implementation. The case of water pollution permits will be used for illustration purposes.

Previous sections argued for the need for recommendations focusing on ways to improve not only AES itself, but also the associated value chains of these services. Here, the term ‘improvement’ is interpreted in a fairly broad sense, and could (depending on the good or service under consideration) refer to an improvement in the quality or the quantity of the service, or derive more value (private and social) from the service, or a combination of these. We focus on the latter, and more specifically on incentives for increasing private sector investment in AES value chains as a specific way of improving aquatic ecosystem service delivery. All of this is done in an attempt to support much-needed growth in the South African AESs economy. Our choice also complements the terms of reference for this WRC study, which argued for analysing ways of ‘taking AES to market’ as one way to improve AES value chains (see Appendix 3). This is a highly complex challenge with the process itself – as the major and important obstacle. This section thus tries to respond to this need by investigating the market-making process associated with AES not only as a first step to engage in this challenge, but to display the complexities and the sheer amount of work required to successfully take AES to market.

After some initial thought, it soon became clear that this process cannot be engaged within a ‘general’ AES context because of the complexity and sheer amount of data required for the market-making process of all the AESs identified within this project. Each AES is unique and requires a dedicated study in terms of its market-making process. In order to achieve an acceptable depth of analysis, it was decided to select and focus on one of the AESs as per the previous work to be used as case study. The water quality control AES was consequently chosen along with one type of water pollution for which we had data (the Dwars River in the Western Cape) as a test case to present a first attempt at designing the market-making process for this AES. The other AESs as identified in this study should learn from this process and be taken further in subsequent dedicated studies.

Some theoretical insights on unlocking private sector investment in AES and the market-making process are given before presenting the actual market-making process for AES. This is done according to the following steps: pre-conditions and valuation, basic characterisation; terms conditions and transactional protocol; monitoring systems; piloting and implementation. But first some theory.

11.1 Some Theoretical Foundations for Unlocking Private Sector Investment in AES

11.1.1 Sustainability as point of departure

Sustainable development could be regarded as a process of creating, maintaining, and managing wealth as defined in its broadest sense. However, society must choose the direction and character of the pathway to be followed towards stainable economic development. Broadly speaking, there are three interlinked pathways that are non-exclusive and complementary:

- Sustainability via technological change whereby the resource and energy intensity per unit output of an economy is reduced (sometimes called the dematerialisation of economic development, if not subjected to the Jevon’s paradox (Jevons, 1866).
- Sustainability via social behavioural change whereby society’s preferences and value systems changes in terms of what is considered to be wealthy, the reason or rationale for living, and the way in which we live (Daly, 1991; Hawken, 1993).

- Sustainability via restoring natural capital, which is defined as any activity that integrates investment in and replenishment of natural capital stocks to improve the flows of ecosystem services, while enhancing all aspects of human well-being (Aronson, 2007).

The first two pathways aim to reduce overall and per capita demand and pollution of natural resources (i.e. natural capital). The third pathway focuses on increasing natural capital to increase the flow of ecosystem services derived from it. There are numerous policy options and intervention packages (taxes, subsidies and regulation) guiding economic development onto these paths. It is, however, the principle of investing in the restoration of natural capital that is important for this project. This principle is embedded within the capital theory approach to economic development (Hicks, 1946) and underpins the idea of a depreciation allowance for natural capital (as used for manufactured capital).

11.1.2 The challenge of rivalry and excludability

Stimulating private investment in ecological infrastructure[2] and the ecosystem services it provides is a unique challenge due to the 'public good' nature of these services. Which is most of time non-excludable[3] and non-rival[4] (see Table 6).

Table 6: Combination of rivalry and excludability (Lipsey & Courant, 1996)

Combination of rivalry and excludability		
	Non-excludable	**Excludable**
Non-rival	Pure public good: Biodiversity, climate regulation	Inefficient market good
Congestible	Congestible public good: Free public beaches, public parks	Toll or club good: Private beaches, game reserves, private ecotourism sites
Rival	Common pool resource: *Ocean fisheries*	Market good: Food, raw materials
Anti-rival	Public good: Genetic information available for public use	Inefficient market good: Genetic information protected by convention on biodiversity

[2] Ecological infrastructure "refers to naturally functioning ecosystems that deliver valuable services to people, such as healthy mountain catchments, rivers, wetlands, coastal dunes, and nodes and corridors of natural habitat, which together form a network of interconnected structural elements in the landscape. Ecological infrastructure is therefore the asset, or stock, from which a range of valuable services flow".

[3] If an ecosystem service is excludable, there is technology or institutions that make it possible to prevent others from using the good or service. No good or service is inherently excludable, although most rival goods can be made excludable through institutions. For example, a t-shirt is certainly rival, but without property rights and enforceable laws, there would be nothing preventing an individual from walking into a store and claiming one. There is nothing intrinsically excludable about the shirt. However, an ecosystem services can be inherently non-excludable. An ecosystem service is non-excludable when it is impossible to create property rights or the costs of enforcement are too high. It would be virtually impossible, for example, to exclude someone from the benefits of climate regulation. A good or service is also non-excludable when the technology or institutions exist to exclude use but property rights are not enforced.

[4] Rivalry is an innate property that cannot be altered by policy or legal institutions. If a good or service is purely non-rival, the use of that good or service by an individual does not have a significant impact on the quality or quantity available to others. However, the quality of some non-rival goods and services can be affected by the number of people using the good or service at one time. These goods and services are considered congestible. For example, a hiker's experience in a nature reserve would not be altered if one other person is in the reserve. Yet, if there were several thousand people in the reserve, the quality of his experience would be diminished. A purely rival good or service, on the other hand, is one in which its use or consumption by an individual precludes use or consumption by another. Commonly purchased goods or services, such as a t-shirt, an orange, or a haircut, fall under this category. Finally, an anti-rival good is one which is enhanced with use by multiple people. Information and some technologies are anti-rival goods. For example, the more people who take a remedy for a contagious disease or use an effective pollution control device, the better off we all are. The marginal value of a rival good is the maximum amount an individual is willing to pay, while the marginal value of a non-rival good is the sum of the WTP of all individuals.

It is this categorisation that determines whether a given payment policy would be an effective mechanism for its provisioning (Table 7). Consequently, such categorisation is handy when deciding upon matching private sector engagement with ecosystem services.

Table 7: Recommended policy approach for different kinds of goods (Lipsey & Courant, 1996)

	Recommended policy approach
Public good	One-time payment by institution acting as monopsony
Market good	Individual payments
Common pool resource	Make excludable through property rights; tradable permits
Toll or club good	Treat as public good; when becomes congestible require one-time payment by individuals
Inefficient market good	Treat as public good; provide incentives for use

However, it should be noted that ambiguity and contextual differences could result in similar ecosystem services that fall in different categories. This differentiation often creates scope for engaging the private sector in ecosystem services conservation.

There is generally little incentive for the private sector to invest in maintaining and restoring ecological infrastructure underlying ecosystem services; mainly because there is generally little or no profit to be made from doing so. Another lies in the difficulty in quantifying the service to be delivered and measuring its delivery in accordance with what has been agreed (security of supply of the service). As such, a market for ecosystem services does not generally exist. Markets require that there is both a willing buyer and a willing seller.

In the case of public goods, there is no willing seller, owing mainly to the characteristic of non-excludability, which means that there are no profits to be made from investing in maintenance and restoration of ecosystems; therefore, no immediate financial incentive to do so. As such, some form of intervention is typically required to promote and enable the private sector to invest in maintaining and restoring ecosystems. Broadly, such interventions usually relate to legislation (e.g. requirements for corporate social investment (CSI)); the emergence and/or development of a business case for investment (e.g. investment provides a business opportunity and/or will assist in realising cost savings or reducing business risks); and/or pressure from civil society (e.g. consumer boycotts).

11.2 PES

While there is much support of the concept of PES (i.e. markets for ecosystem services), and while the value of investing in natural capital has been demonstrated (Admiraal et al., 2013; Azqueta & Sotelsek, 2007; Costanza et al., 1997; Daily et al., 2009; Waage, 2007), capturing this value and operationalising it in suitable markets remained uncommon. This is mainly due to the many uncertainties embedded within the value chains of ecosystem services, which create risk that discourages market development. Market developing initiatives focused on increasing private sector investment in PES will consequently need to accommodate these risks to allow such investment that will in turn stimulate and grow the market for ecosystem services (Trabacchi & Mazza, 2015). This risk-averse nature of the private sector is therefore central towards designing a market mechanism that will incentivise private sector investment in PES. Seeking an answer for this dilemma requires a thorough understanding of the basics of private sector functionality and the tension between public goods and the private sector (Bernholz, 1997; Coase, 1937; Coase, 1960).

Government has launched the Green Fund to engage this challenge by providing catalytic finance to partly account for the risk to allow private sector to invest in green initiatives. Response has been good except for the so-called “environmental and natural resource management” window, where comparatively few applications were received (Audouin & De Lange, 2016). The question could be asked how to incentivise private companies to invest in protection of natural capital (i.e. ecological infrastructure) to ensure sustained ecosystem services delivery.

In theory, private sector investment could be incentivised via three avenues:

i. Enforced legislation via command-and-control measures.

ii. Pressure from civil society (social shame and consumer boycott).

iii. A business case (financial viability), which could be subdivided further in the following:

- Investment in ecosystem services provides a business opportunity.
- Investment in ecosystem services realises cost savings.
- Investment in ecosystem services secures operations (licensing for mines).
- Investment in ecosystem services appreciates the value of the assets belonging to the company.
- Investment in ecosystem services decreases the company's risk profile.

All three of these avenues could be serviced via the market, albeit in different ways:

- Compliance markets, that is, public regulation requires the payment for using the ecosystem service (e.g. mandatory carbon emission trading for certain industries).
- Government-mediated markets, where government is the intermediate party collecting payments from users and distributing the funds to service providers (e.g. PES markets based on water services).
- Voluntary markets in which companies voluntarily decide to compensate their impact on the environment by purchasing compensatory credits (e.g. voluntary carbon emission credits).

Although all three could be used, last-mentioned (i.e. a business case for voluntary markets) is the focus of this report since it has the biggest potential to be self-sustaining in the long term (i.e. does not require government funding to ensure its functionality once it is up and going). However, within voluntary markets, private sector will challenge the link between investment in ecosystem services and their own financial viability in several ways:

- How does investment in ecosystem services affect the company's decision-making regarding capital?
- How does investment in ecosystem services affect or enable more efficient operations?
- How does investment in ecosystem services affect the risk profile?
- Does investment in ecosystem services address consumer/client/customer needs?

All these questions draw information from the business case of investment in ecosystem services since the private sector will only invest voluntary if such investment makes business sense, i.e. improve their financial viability. The business case for ecosystem services need to improve the financial viability of a company and can be done via different pathways:

- If investment in ecosystem services provides a business opportunity where such investment could **<u>stimulate new demand</u>** for a new product/service, which then increases the company's turnover.
- If investment in ecosystem services secures, sustains or **<u>reduces costs</u>** of key natural resource-based inputs required for business operations, e.g. such investment could realise a production cost saving if such investment produces inputs of superior quality/quantity for the same direct cost or maintain quality/quantity for a decrease in direct cost. For example, clean water from a wetland that does not need to go through expensive filtration processes.

- If investment in ecosystem services **secures a licence to operate** by managing fragile relationships.
- If investment in ecosystem services **appreciates the value of assets** belonging to the company.
- If investment in ecosystem services **decreases the company's reputational risk** via 'green' branding. Here it is often the case that a price premium for a product or services could be justified if the buyer of such product or service has a WTP for a 'green' product/service. Such WTP derives from various incentives, of which social status and self-identity are examples.

All these avenues represent incentives for private sector investment in ecosystem services because they all improve the financial viability of the company. However, the financial benefit of ecosystem services can only be realised if it is captured through a transaction within a market or similar agreement. Examples of such markets include:

- Market for carbon sequestration.
- Market for water provisioning.
- Market for water treatment.
- Market for soil stabilisation/erosion/sediment control.

Above-mentioned markets could take different forms:

- Public sector buying ecosystem services from public sector.
- Public sector buying ecosystem services from private sector.
- Private sector buying ecosystem services from public sector.
- Private sector buying ecosystem services from private sector.

All these markets require different sets of rules, thus creating an enabling environment with legal backing to lower transaction costs that will facilitate market activity (i.e. transactions). If this is not in place, private sector will not invest because the sector is risk averse and will not compromise its financial viability. This means that the private sector will be slow to engage unless they receive security/assurance to hedge against risk. Consequently, the terms and conditions for transactions for these markets are yet to be developed and tested. One cannot therefore expect an active PES market in South Africa. The challenge in agreeing on the terms and conditions of such a transaction therefore stand in the way of demonstrating the value of PES to the private sector, and this is where catalytic finance (as provided by the initiatives such as the Green Fund) could play a vital role. Furthermore, one needs to account for the fact that most ecosystem services are public goods that are non-excludable and non-rival (as explained earlier). These two properties pose special requirements to the terms and conditions of markets for ecosystem services, which need to be accounted for when drafting the rules of the game.

11.3 Value Chains for Ecosystem Services and the Market-making Process for PES

A pragmatic way to illustrate this dilemma and to work towards a solution is to use the concept of a value chain, or rather an ecosystem services value chain. We believe that the process is beyond the point where private sector needs to be convinced regarding recognising the monetary value of ecosystem services. The concept has been illustrated repeatedly in literature. Taking stock of the potential value (i.e. potential market value) has also been demonstrated by several studies (MA and TEEB initiative etc.). However, capturing this value within a market where buyers and sellers of ecosystem services employ agreed-upon terms and conditions to make a deal (i.e. designing the product or service and the conditions of sale that will come into play when changing ownership during the transaction) remains a challenge because such conditions have not been developed for South

Africa (i.e. rules of the market have not been set) and the potential buyers of this value consequently remain scarce.

The associated transaction cost (i.e. effort required to clinch a deal) therefore remains too high for most companies to make it worthwhile (i.e. to realise a nett value). Engaging these transaction costs should be the main focus area of government support (e.g. the Green Fund) to incentivise private investment in ecosystem services (Blignaut & Van der Elst, 2014). Once such transaction costs have been reduced, the PES value proposition could be realised by market players participating actively. This process will be a function of the demand and supply of ecosystem services – a process that is determined by changes in user preference, which cannot be rushed. One needs to remember that markets are indeed anthropogenic constructs reflecting the perceptions, will and intent of people, which operate according to agreed-upon rules and conventions. South Africans are still catching on to the idea of PES (the idea of restoration and PES are still relatively new to the South African society). Changing these perceptions takes time.

The value chain for ecosystem services comes in particularly handy for designing terms and conditions that will facilitate market transactions, since the value chain points out areas where the disconnect between sellers and buyers of ecosystem services lies (i.e. it creates the context for the intervention). The requirement for specificity within the PES market-making process becomes apparent here since the economic and biophysical/spatial characteristics of ecosystem services differ. Terms and conditions need to target that specific area of the chain to facilitate a transaction (context-specific terms and conditions) (Pirard, 2012). These points of disconnect will be at different places for different ecosystem services value chains. Initiatives aimed at incentivising private sector investment need to be structured in such a way as to seek out and focus on these points of disconnect, thus a market-making process is required.

It should be noted that while this market-making process is onerous, complex and costly, it also requires patience, a universal belief in, and a concerted effort of the players in the value chain to capture the demonstrated value. An example would be the market for carbon credits from restored natural veld, which requires verification and validation processes, payment agreement/protocols, validation of brokers, consequences of non-delivery etc. This kind of information needs to be based on evidence and formalised in the market terms and conditions. Some experience from abroad (Appendix 2) could be helpful, but given the differences in contexts, pilot projects are required to test and demonstrate the workings of, for example, a carbon credit market in South Africa on pilot scale before the private sector will engage.

Government's primary role in this market-making process should not to be a market player (i.e. supplier or buyer of ecosystem services) but should focus on institutional protection and safeguarding to protect the parties involved in the market-making process (i.e. decreasing their risk). Such protection thus not only becomes an insurance policy and mechanism against premature market failure, but also creates a platform to develop trust and a transaction record of cooperation (i.e. evidence-based outcomes) to learn from and from where to launch further transactions. The role of government should be to provide institutional support for the market-making process that will allow private sector engagement.

South Africa has a significant amount of natural resources and has the potential to be a major player in the PES industry. However, it is unlikely for South Africa to take up its position in the global market without these barriers to entry being addressed. Markets for PES cannot operate on the same rules and regulations as markets based on private goods. There is a need to draft PES-specific terms and conditions and to test these on pilot scale. The challenge is that markets for PES have different requirements for product/service characteristics, market player arrangements and the terms and conditions of transactions than mainstream markets focused on private goods. The complexity of these markets necessitates an enabling context, particularly for private sector entrants, many of whom are interested in mitigating risks. The entry barriers and the need to facilitate the market-making process should be prioritised for high-level government intervention (e.g. the Green Fund), to assist in the market-making process. This work will need to account for the following:

- Misalignment between PES, industrial policy and the structure of the financial system.
- The legal and regulatory framework needed to support payments schemes.
- Market terms and conditions necessary to guide individual transactions. Ambiguity will be kept to a minimum if definitions for the terms and conditions of transactions are clear.
- Technical services necessary to implement and monitor trading.
- Encouraging multi-stakeholder cooperation is critical because of the public good nature of ecosystem services. The public good nature of ecosystem services implies that the market-making processes for PES require public regulation or public-private partnerships as the basis on which to build a business case attractively enough to stimulate investment from the private sector.
- Increased information sharing will be required to stimulate the market.
- Innovative financing models will need to be designed. While investment fund managers have a limited understanding of PES, leaving PES being excluded as an investment option, the market have the potential to be the future driving force of economic growth.

There are very few (if any) examples in South Africa where PES has successfully been brought to market. This is mainly because South Africans is not ready for such markets (user preference as explained above) and because PES market-related research has focused on assessing the value of ecosystem services with comparatively little funding been made available to design and analyse the actual market-making process of these ecosystem services. Examples include:

- **Water quantity:** The link between clearing invasive alien plants and water run-off/base flow has been analysed, but is still not always certain. Empirical evidence to confirm these relationships in a broader hydrological context is required to advance this market. This can only be done by having well-monitored projects, which could help to calibrate the hydrological models.
- **Water quality:** Several theoretical studies have been done to explain the water quality regulation function of wetlands. However, pilot studies displaying the relative difference in performance of wetlands vs artificial filtration is required to display the value of these ecosystems.
- **Drought and flood attenuation:** During times of such extreme events, the quality of the stock of natural capital acts as a buffer, mitigating the effects of such an event if the natural capital stock is of high quality. Conversely, if the quality of the natural capital stock is bad, the effects of such extreme events are deepened. Investment in natural capital and the restoration thereof could, therefore, be seen as a form of insurance policy, more so in the wake of the likely effects of climate change. This, however, is not yet a readily accepted form of insurance policy, yet it should be and work towards this end is required.

Initiatives such as the Green Fund (mentioned earlier) perhaps need to follow a more structured and focused approach when drafting the scope and terms of reference for solicited research focused on the market-making process of specific ecosystem services. Such research needs to identify gaps within the value chains (i.e. areas of disconnect or barriers as mentioned above) of these ecosystem services within specific spatial contexts and then design terms and conditions to bridge the gap. The research will also need to address transactional protocol for the **specific** PES since the process and transaction conditions will differ between PES. Buyers and sellers in this market are not monolithic, which implies that each market for PES differs in terms of barriers and motivations for its market actors. The market-making process will also need to consider the supporting services and agents required for the market in question. The functionality of these markets then needs to be challenged and tested by means of pilot-scale studies to illustrate the relative efficiency of the terms, conditions and protocol in capturing the financial value of the good or service. Such pilot-scale successes could then be upscaled by loans at favourable terms to support restoration projects, and projects that would develop PES.

11.4 Conclusions from the Theory

So, what could be learned from the theory? It is should be clear that the complexity of markets and PES necessitates an enabling context, particularly for private sector entrants, many of whom are interested in mitigating risks. In summary, the barriers to entry and the need to facilitate the market-making process should be prioritised for high-level intervention, through initiatives such as the Green Fund (or similar government initiative), to assist in the market-making process. The remainder of the report applies the above-mentioned theory by presenting an example of the market-making process of a particular service.

12 A MARKET-BASED APPROACH TO MANAGE POLLUTION

In a free market economy, private (firm and individual) production and consumption decisions are based on trade-offs between WTP and WTA private costs and benefits, which are reflected in market prices. According to neoclassical economics, the ‘invisible hand’ of the market is assumed to ensure that these private decisions will lead to socially optimal outcomes, such as optimal levels of production and pollution (Goodstein, 2008). However, people tend to give more prominence to private costs and benefits, resulting in choices that are not always socially optimal. When the costs of producing a product or the benefits from consuming a product spill over to people who are not involved in the consumption or production of the good, an externality occurs. Such external impacts are thus unaccounted-for costs (such as pollution) and benefits (such as education). The market fails to accommodate these impacts in the market price, i.e. the market ‘fails’. Consequently, market prices often fail to adequately reflect the full *social* costs and benefits associated with these goods or services due to the existence of externalities. Subsequently, the levels of production and pollution will not be socially optimal because the trade-offs are not reflected accurately. With negative externalities specifically, social costs exceed private costs (making them per definition negative), such that too much of the pollution activity will be undertaken relative to the socially optimal amount. Pollution is therefore an example of a negative externality (an external cost of production or consumption) where market prices provide incentives for too much environmentally damaging behaviour. ‘Internalising’ such externalities therefore become necessary to readjust prices in such a way that the negative impacts of pollution will be considered by the polluters. However, such readjustment requires an estimate of the monetary value of the impact of pollution. Such valuations not only enable this internalisation process, but can also be used to compare different pollution mitigation strategies within a particular area and to a lesser extent, similar areas. Such valuations can also enable the use of more advanced policy instruments such as tradable pollution permits. These permits are a form of market-based governance that seeks to change pollution behaviour by changing price signals to which economically driven actors are expected to respond in their own self-interest. In this way, markets may harness the decentralised power of individual decision makers to achieve policy objectives set by government (who also design the terms, conditions and transactional protocol for the market and regulate its subsequent operation).

South African policymakers have many more environmental protection tools at their disposal than they did 20 years ago when so-called command-and-control mechanisms to regulate unwanted behaviour was the preferred approach. Although this approach was effective in some areas, it proved to be costly and difficult to enforce for water pollution because of the high level of monitoring required by methods based on this approach. Market-based mechanisms provide alternatives to command-and-control mechanisms. These could take the form of subsidised reforms, taxes to account for social costs, or establishing markets where pollution permits (i.e. the right to pollute a certain amount per time period) can be traded (see Section 11.2). The latter is of particular interest for this study as it not only aims to limit pollution at an optimal cost to the polluter, but also create an incentive for companies to reduce pollution further (relative to their entitlement), since it becomes possible to sell the difference to willing buyers (i.e. a firm who cannot meet their pollution targets) for a profit (refer to Section 11.2 for other reasons why companies might be interested in reducing their pollution). Although such trading happens within a predetermined pollution standard, it can lower the cost of compliance while realising pollution prevention benefits.

Market-based systems thus capitalise on the power of the marketplace to reduce pollution cost effectively and use economic incentives to promote conservation (David, 2003). It does however, require an innovative market-making process to design the necessary terms and conditions that will allow fair trade. Nonetheless, there is recent general and empirical evidence of success in literature (Goulder, 2013; Newburn & Woodward, 2012; Ribaudo & Gottlieb, 2011; Shortle 2013; Van Houtven et al., 2012; Wiedeman, 2001), all within the developing world, which is why this kind of work is quite novel for South Africa.

The market-making process draws heavily from appraising the monetary significance of the pollution problem that underlies water quality control related AES, i.e. water pollution is the result of not having the AES. We need to have a monetary valuation of the pollution impacts when the AES is compromised. Because of the sheer amount of data required for such an exercise, we have decided to do the design for a study area and pollution problem for which the team have decent field data. ***Consequently, we focused on nutrient enrichment and the consequent impacts of filamentous green algae pollution on commercial agriculture in the Dwars River, Western Cape*** (De Lange, 2014) in South Africa. Here, algae often deplete sections of rivers from oxygen leading to eutrophic conditions, fish kills, and an increase in the operation and maintenance cost of irrigated agriculture (it should be noted that nutrient enrichment is also a problem downstream of the Kouga Dam of Baviaanskloof). One of the mains reasons for focusing on agriculture is because the market can potentially engage the so-called ‘non-point’ pollution challenge associated with agriculture (where traceability of pollution to polluters is impossible); which is a major problem for this sector.

We therefore present another study area for the market-making process in the next section before discussing the methodological approach for the valuation and presenting the pollution impacts and the calculated monetary value of the impacts of such pollution. The market-making process is then presented along with a first attempt at the initial price setting and description of terms, conditions and transactional protocol for a market for pollution permits specifically for this kind of pollution. It is anticipated this work will give a good indication of the complexities involved and sheer amount of work required to take AES to market.

12.1 Study Area

The Dwars River (Figure 29) is a major tributary of the Berg River, which is a major source of water supply to the Cape Town metropolitan area. It is an area with high rainfall on the peaks (>3000 mm/yr) but with very steep rainfall gradients. Although the area is about 10% of the surface area of the relevant quaternary catchment, it yields 24% (approximately 23 million m^3/yr) of mean annual run-off of quaternary catchment. The average rainfall is 877 mm/yr.

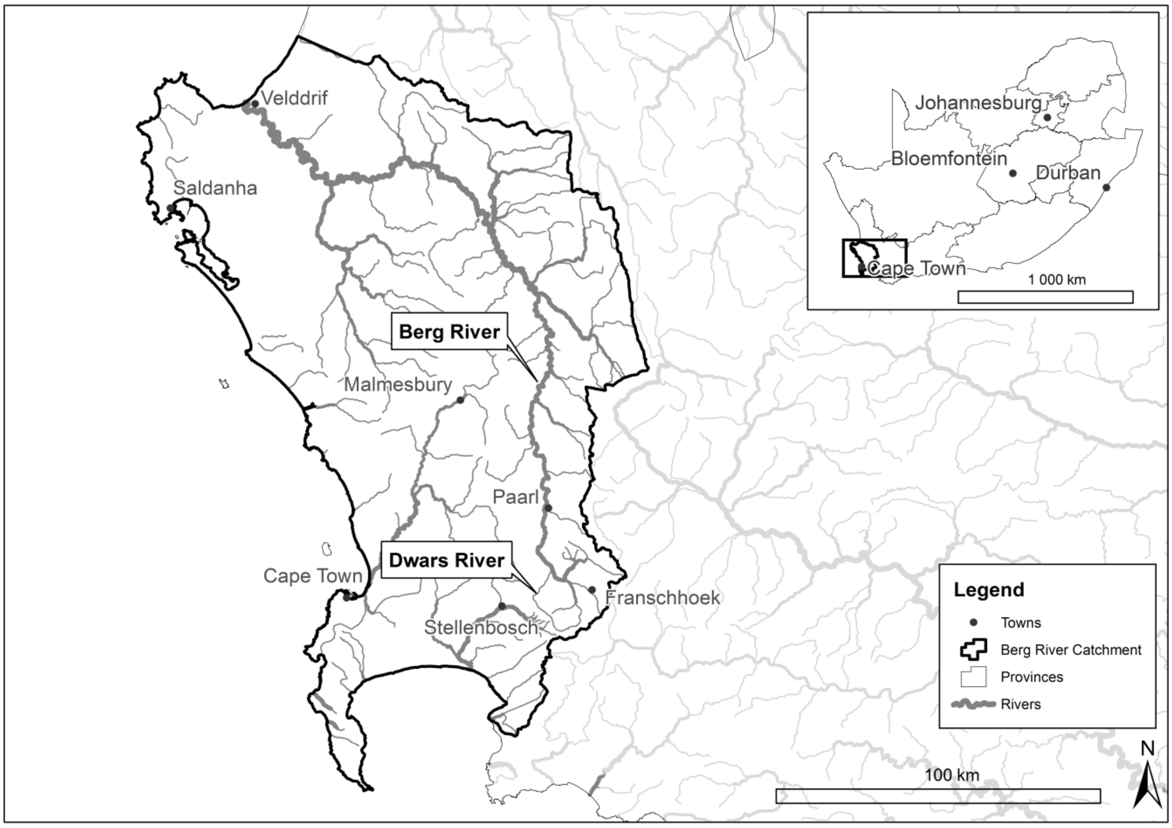

Figure 29: Approximate location of the Berg River catchment study area

12.2 Approach for Valuation Pollution Impacts

We applied a production function approach (Birol et al., 2006; Brouwer & Pearce, 2005; Glazyrin et al., 2006; Pearce, 1993; Pearce, 1994) to estimate the monetary value of the impact of filamentous algae on commercial agriculture. We focused on the impacts of filamentous algae growth on farm profitability, which relied on detailed information on the impact(s) and the extent of the impact(s) of filamentous algae on farming practice. The basic valuation procedure included:

- Representative crop selection and construction of a typical farm profile for each representative crop.
- Description and quantification of the impacts of filamentous algae on the cultivation practice of representative crops.
- Valuation of the impact of filamentous algae.
- Aggregation and extrapolation to the level of the water user association.

Commercial agriculture in the Dwars River focuses (in terms of hectares) on deciduous fruit and viticulture (DFPT, 2013). We could not include all deciduous fruit, which is why plums was taken as the representative deciduous fruit crop since it represents 70% (307 ha) of the area under deciduous fruit. There is also approximately 355 ha of irrigated wine grapes in the study area (all vines were included) (SAWIS, 2013).

Prominent farmers of the representative crops explained the impacts of filamentous algae on their business in terms of the impacts on their cultivation practice. It was assumed that filamentous algae are always present in the water. A difference in concentration levels is considered the distinguishing factor determining any mitigation strategy and hence cost implications and consequent profitability impacts. Farmers were distinguished between a ‘heavy’ and a ‘normal’ filamentous algae load scenario.

The cost implications of the impacts of filamentous green algae were determined by systematically accounting for the cost variables involved in mitigating (i.e. managing) the impacts of filamentous algae. This process was done in close collaboration with the farmers because mitigation strategies for filamentous algae differ between farms. Steps in their algae mitigation process were described while noting the cost implications. The cost was systematically captured in a spreadsheet to calculate the total direct cost of the on-farm pollution mitigation process.

We structured the cost impacts according to the crop enterprise budget (cost structure for standard cultivation practice) for each representative crop. Supplementary industry data was obtained from Hortgro, SAWIS, VinPro, Nulandis and Kaap-Agri.

12.3 Pollution Impacts

Filamentous green algae thrive under eutrophic conditions due to nutrient enrichment from raw or partially treated sewage, agricultural effluent, and other forms of phosphorous-rich pollutants (Oberholster & Botha, 2011; Oberholster et al., 2013). Although filamentous algae pose no direct threat to crops, algae affect the operational efficiency of irrigation systems. Therefore, algae affect the operation and maintenance costs of irrigation infrastructure, which increase downstream. It should be noted that although pollution loads could vary during the year, the impacts affect farmers during the irrigation season, which starts in the third week of October and lasts until the second week of March for Dwars River.

There is no bulk supply infrastructure in Dwars River (farmers draw water directly from the river). Most farmers were aware of the direct relationship between bioavailable phosphate and filamentous algae. They believed that filamentous algae affect farm profitability directly via increased irrigation costs. Filamentous algae not only obstruct and clog strainers, intake valves and manifolds, but also place a higher load on impellers and bearings of pressure pumps, while decreasing the delivery rates. This decreases the volume per pump-hour. Micro-jets and dripper lines are also clogged leading to uneven and thus suboptimal water application in orchards, which affects crop yield if not mitigated. More frequent cleaning of foot valves, intake manifolds, filter banks nozzles of micro-jets, centre pivots and drippers are therefore required. It also implies more frequent dosage of on-farm balancing dams with copper sulphate and sometimes hydrogen peroxide in irrigation systems (Smith, pers. comm., 2014). Some farmers argued that hydrogen peroxide decreases the service life of all plastic components since it becomes brittle.

12.4 The Cost of Eutrophication

The standard practice to manage the impacts of filamentous green algae was clearly more frequent cleaning of irrigation systems, which provided the basis for a cost estimate on the impact of algae. However, any irrigation system requires a minimal amount of cleaning and flushing prior to, and during the irrigation season, hence the need to differentiate between costs related to standard practice and costs related to high filamentous algae loads. It was consequently decided to distinguish between a ‘high’ and ‘low’ algae load scenario in order to differentiate the cost.

Farmers were asked to systematically describe the steps taken in their standard protocol of dealing with filamentous algae within a typical pump station on their farm. This allowed a differentiation between a high and low load scenario and therefore cost differentiation. However, as farmers neither quantify nor keep a record of the volumes of filamentous green algae removed, the exact definition of high remains subjective. The following assumptions and input variables were used:

- The irrigation season lasts 20 weeks (i.e. 120 days).
- Farm labour cost is R15 per hour.
- A typical pump station serving 10 ha consists of a strainer, foot valve, 3.3 kW pump, sand filter and a disc filter.
- Electricity cost is R1.20 per kWh.
- Approximate labour input for cleaning a strainer and foot valve is 15 minutes.
- Approximate labour input for cleaning a disc filter is 10 minutes.
- Approximate labour input for backwashing a sand filter is 10 minutes.
- Approximate electricity input requirement for backwashing is 0.55 kW.
- Traveling time to and from pump station is 20 minutes.

Table 8 presents the difference between operating and maintenance activities for typical low and high algae load scenarios in Dwars River.

Table 8: Operating and maintenance activities for low and high algae load scenarios in Dwars River

Activity	Low load scenario	High load scenario
Service pumps	Every third year	Every second year
Replace filter sand	Every third year	Annually
Clean strainers, disc filters and sand filters	Once a week	Daily
Required man-days per week to clean micro-jets and drippers during irrigation season	1	7

The service cost of a pump averaged R9000. This implies that the annual service cost for a pump under low load conditions is approximately R3000 per year, and R4500 per year under high load conditions, which is a difference of R1500 per year. Filter sand costs R1875 per filter, which implies an annual cost of R625 for low and R1875 for high load conditions, i.e. R1250 per year difference. Table 9 represents the labour cost per activity.

Table 9: Calculated cost per pump station in the Dwars River

	Low algae load conditions		High algae load conditions		Difference (R)
	Total time during irrigation season (minutes)	Rand value (R)	Total time during irrigation season (minutes)	Rand value (R)	
Labour for cleaning micro-jets	9 600	2 400	67 800	16 800	14 400
Labour for cleaning disc filter	200	50	1 400	350	300
Labour for backwashing	200	50	1 400	350	300
Transport time	400	100	2 800	700	600
Servicing of pumps		3 000		4 500	1 500
Filter sand		625		1 875	1 250
Electricity for backwashing	11 kW	13.20	77 kW	92.40	79.20
Total additional cost due to eutrophication per pump station per year (10 ha)					18 879.20
Total additional cost due to eutrophication per hectare per year					1 887.92

The total estimated cost attributable to eutrophication is thus estimated at R1887/ha per year for the representative crop (plums). Given an estimated 307 ha of deciduous fruit in the study area, and assuming that all farmers struggle with filamentous algae, we could infer that the above-mentioned R1887/ha per year (see Table 9) translates to R579 591 per year for deciduous fruit in the study area. Given that the pre-harvest production cost for plums is R61 880 per hectare (DFPT, 2013), the cost of filamentous algae pollution represents approximately 3.1% of pre-harvest costs.

Although vines use drip irrigation while deciduous fruit uses micro-jets, the labour component for cleaning these two types of irrigation systems are essentially the same. Consequently, the mitigation strategy and cost per hectare for managing filamentous algae is comparable. Given an estimated 355 ha under vines in the study area, the cost of filamentous algae for the wine grapes is estimated at R670 400 per year (assuming comparability in terms of the filamentous algae problem across farmers). Pre-harvest production cost for wine grapes is R35 739 per hectare (Van Niekerk, 2013), which implies that the filamentous algae problem represents 5.2% of the pre-harvest cost. It could thus be argued that the filamentous algae problem costs the vine and deciduous fruit industry approximately R1.2 million per year in the Dwars River.

The information generated thus far not only creates an overarching ‘management budget’ for filamentous green algae in the study areas (i.e. any overarching filamentous algae management strategy will be considered worthwhile from a financial perspective if the cost to mitigate the filamentous algae impacts is less than R1.2 million per year in the Dwars River), but also generates valuable information for the market-making process for tradable water pollution permits.

In the next section, we present some of the intricacies of the market-making process for this kind of permit by discussing the theoretical foundations of the concept, presenting a methodological approach for designing such a system, including some of the major components of the process. The focus continues to be on filamentous green algae within the same study area.

13 THE MARKET-MAKING PROCESS FOR WATER POLLUTION PERMITS

Tradable pollution permits (also called cap-and-trade) is a market-based approach using economic incentives to reduce pollution. It is based on the polluter-pays principle (Glazyrin et al., 2006; Kraemer & Banholzer, 1999; O’Neil, 1983; O’Neil et al., 1983, Ribaudo & Gottlieb, 2011) and aims to impose a cost on pollution, or generate a reward for pollution abatement. Individual participants may then trade to their mutual advantage within the constraints as set by the rules of the market. Although the transfer of permits is referred to as ‘trade’, in effect, the buyer of a permit pays for the right to pollute, while the seller receives compensation for letting go of such right. Thus, in theory, those who can reduce pollution most cheaply will do so, achieving pollution reduction at the lowest cost given the cost structure of the two parties (i.e. not necessarily the optimum solution for society). It is therefore not only a way to harness market incentives for controlling pollution, but also a transparent process that promotes fair mitigation of pollution while gradually identifying non-point polluters if the buyers of permits are indeed polluting.

A pollution permit scheme thus uses market-based incentives geared towards making pollution an expensive activity that will not only create an incentive to reduce pollution directly, but will also create incentives to adopt cleaner production and consumption activities. It also tends to lower pollution regulation cost by leaving decisions regarding how to reduce their pollution up to the polluters. It is assumed that polluters will choose the least expensive way (given their unique circumstances) to mitigate pollution regulation.

Challenges associated with this approach include the complex nature of the required terms, conditions, transactional protocol and operating rules that will facilitate transactions (i.e. market activity). These need to account for potential inflationary impacts, problems of monitoring and enforcement, hot spots (high local concentrations of pollutants), thin markets and exposure to possible monopolistic market power. The remainder of the report focuses on this market-making process presenting some headway been made to engage some of these challenges.

13.1 Some Relevant Theory on Pollution Permits

Morgan and Wolverton (2008) present an overview of water pollution trading programmes and once-off offset agreements in the United States of America (USA) while Ribaudo and Gottlieb (2011) follow by focusing on key challenges and lessons learned (Shortle, 2013). Although not all of this is relevant to the South African context, the basic premise of a tradable permit still applies.

Pollution permits are in effect a legal right to pollute a certain amount within a specified time period (the permit life). If the polluter produces less pollution, they can sell the right to another polluter who might not be in a position to meet the required pollution targets. A market for the right to pollute is therefore created. Those who generate significant pollution or cannot comply with pollution targets can buy permits from those who pollute less. If a polluter is unable to find a willing seller, they will face a pollution tax from the state, which could be equal to the current value of a permit plus a premium equal to a stipulated fine.

The key difference between a pollution tax and a permit mechanism is that with a tax, the price is established ex ante and the extent of the pollution reduction is determined by the level of compliance of the polluters. With a permit, the pollution target is set ex ante, and the pollution reduction is then priced based on the cost of mitigation.

Tradable pollution permits therefore not only creates a strong incentive to pollute less, but the flexibility of a market mechanism also allows polluters to use the most affordable pollution compliance strategy given their internal marginal pollution abatement costs and the market price for permits. In theory, a polluter's individual decisions should then lead to an economically efficient allocation of permits and lower overall pollution compliance costs as compared to command-and-control mechanisms. Over time, a cost-effective outcome will be achieved by a well-functioning market regardless of the initial allocation of ownership of the permits (see the Coase theorem, Coase, 1960). In theory, tradable permits could have the following benefits over command-and-control mechanisms (Boyd, et al 2003):

- Address environmental impacts more directly by setting quantifiable physical limits on pollution, and implement strict monitoring to ensure compliance.
- Allow polluters greater flexibility in the choice of means to comply.
- Decrease the overall cost of compliance by encouraging polluters who can comply more cheaply to do so first, while allowing those with higher costs to comply by opting to buy additional permits.

There are however, several challenges associated with pollution permits:

- Permits can (as with pollution taxes) have inflationary impacts when polluters can pass the cost increases on to consumers.
- It is challenging to determine the number of permits to be issued initially because the level of pollution varies over time. Too many permits can result in a very low permit price, which reduces the incentive that permit-liable polluters have to cut back on pollution. Too few permits can result in excessively high permit prices leading to 'thin markets', i.e. little market activity.
- Rich polluters can simply buy themselves out of trouble, thereby weakening the desired effect of pollution reduction.

- When permits are issued free of charge since it can create incentives not to cut pollution. Therefore, most permit schemes auction the initial allocations with an independent valuation such as a reserve price in place. However, auctioning initial permit allocation could have more political opposition than free handouts.
- Adding to the previous point, even if permits are paid for, it still has a dampening effect on incentives to find innovative answers to reducing pollution because the cost of innovative answers could be higher than the permit price.
- The nature of the pollutant plays an important role in the design of the mechanism. For example, the market for carbon dioxide is perhaps the most well-known because it acts globally, thus its impact on the environment is generally similar across the globe and the location of the polluter does not really matter from an environmental standpoint. This uniformity makes for easier design of the terms and conditions for the carbon market. A pollutant with more localised impact (such as water pollution) requires more localised terms and conditions because its impacts change and differ across locations. Terms and conditions for these markets are therefore not necessarily universally applicable. Also, the same amount of pollutant can exert different impacts in different locations. This implies that the location of the pollution matters, which is known as the ‘hot spot’ problem.

All the above and the fact that water is a classic public good (i.e. its non-excludability includes free-ride problems that lead to a tragedy of the commons) with complex non-point characteristics, makes a water pollution permit scheme an extremely complex challenge to design and manage effectively. It is therefore to be expected that tradable pollution discharge permits are among the most challenging market-based instruments for water management and pollution control in terms of both their design and implementation. Consequently, few examples of successfully and functioning tradable water pollution permit schemes exist. Many countries have considered these schemes, with several reaching advanced stages in the development, stopping short of implementation (Nishizawa, 2003; NSW-EPA, 2003; Wie~~i~~deman, 2001). Thorough design is a necessary but not sufficient factor for success as the process also requires political support and buy-in from stakeholders as such a system could require substantial changes (reform) to existing regulatory frameworks and institutions. This is where thorough piloting of the terms and conditions becomes valuable since it increase stakeholder buy-in via an opportunity for consultation between designers and participants, which is vital for making the benefits of trade obvious and for increasing the acceptability of the scheme.

Based on the above-mentioned general framework, the remainder of the report presents some headway that has been made in terms of the market-making process for nutrient-related pollution permits in our study area. It is uncharted terrain for South Africa, with no standard procedure. A focused effort was made to draw on current sources of information and to identify information gaps for each of the following steps:

1. The basic characteristics of permits, such as the physical parameters of standardised measurement protocol (water pollution permits should be based on loads (the concentration multiplied by the volume over a specified time period), rather than on concentrations only), establishing the pollution cap (reflecting the maximum amount of pollution the river can safely absorb), number of permits to be issued, terms and conditions of transfer, the geographic scope of the scheme and eligibility criteria of participants, were captured.
2. Design the terms and conditions and transactional protocol including the parties involved in a transaction (buyer and seller only; through agents/brokers; at an exchange, or carried out under the auspices of an administrative authority). This step also considers the compatibility of the proposed system with existing legal, regulatory frameworks and institutions to keep transaction costs down (Smith, 1999). It should also consider ways to ensure temporal and spatial flexibility in the system.

3. Design the monitoring system to track performance and establish credibility in the system. The monitoring system would therefore need to be designed to provide data that will stand in court for resolution of potential conflict.
4. The terms and conditions should be tested on pilot scale and the expected ability to meet changes in environmental and resource requirements. This will include the ability to respond to changes in the boundaries of the system, such as an expansion of the physical coverage of the scheme. However, it should be kept in mind that once implemented, these terms and conditions should not be unexpectedly revised since it might harm investment confidence and may depreciate the value of permits. Hence the need for thorough piloting.
5. Implement via the initial allocation of permits, including consideration of the mechanism of initial allocation (free handout by registration; by application criteria; or by auctioning), and if not free of charge, consider the initial price.

13.2 Towards a Water Pollution Permit in the Study Area

Following the above-mentioned five steps, we took the objective as '*to initiate the establishment of a tradable pollution permit system in the study area'.* The absolute level of pollution needs to be reduced and it was expected that such a system will enable more transparent and fair (i.e. the polluter pays) management of pollution, while gradually identifying non-point polluters (a common phenomenon in agriculture) that could allow a more focused and efficient approach for command-and-control measures on these polluters.

13.2.1 Permit design

Define the unit of measurement for the permit

In contrast to water use rights, which can be expressed in time-based volumetric units (Armitage et al., 1999; Louw, 2002; OECD, 2001; Van Heerden et al., 2008), water pollution permits have to cope with a number of pollutants, all with distinct effects on ecosystems. Water pollution permits should not only account for several substances, but also for the location of the discharge. Furthermore, fluctuations of these variables are determined by area-specific natural and anthropogenic variables. Filamentous green algae hold true to this complexity with several factors determining its occurrence. For example, it requires flow velocities lower than 0.8 m/s, total phosphorous of 0.03 mg/ℓ and total nitrogen of 0.5 mg/ℓ (Summers, 2008).

Furthermore, water hardness (i.e. the magnesium and calcium carbonate content of water) and alkalinity (i.e. the capacity of water to neutralise acids) controls the availability of phosphorus uptake and hence the growth rate (Summers, 2008). Summers (2008) specifically noted that "calcium and magnesium ions act to control the availability of phosphorous for algae uptake". The association of algal growth with hardness and alkalinity is because bicarbonate ions increase the supply of carbon dioxide for photosynthesis (Smith, 1950) and carbonate-bicarbonate buffering that controls pH values (Patrick, 1977).

It could therefore be argued that alkalinity and hardness are two important determinants of the uptake of phosphorous, and consequently for filamentous algae growth in river systems. While these parameters (see Table 10) affect the level of algae growth, they also provide focus points for pollution mitigation strategies, i.e. mitigation strategies could focus on these parameters to affect the level of filamentous algae.

Table 10: Measurement parameters for alkalinity and hardness in river systems

	Measurement (mg/ℓ)	**Range for ideal conditions favourable for benthic algae growth (mg/ℓ)**
Alkalinity	$CaCO_3$	75–200
Hardness	$CaCO_3 + MgCO_3$	100–150

Although alkalinity and hardness could be treated in theory, it is seldom the case for river systems. Treatment of the river system rather focuses on phosphorous and algae (Wehr & Sheath, 2003). Phosphorous could be managed in several ways:

- Diversion and advanced waste water treatment.
- Detention basins and wetlands.
- Alum dosage.
- Dredging.
- Aeration, which reduces the amount of phosphorous being oxidised in the sediment and released into the water column.
- Hypolimnetic withdrawal via removal of high nutrient load bottom waters by pumping it to larger water systems, although this just relocates the eutrophic problem.

Direct control of filamentous algae:

- Direct harvesting by hand/or mechanised equipment.
- Biological control and bio-manipulation: filamentous algae predators will differ depending on their species and range from phytoplankton species to piscivorous fish, while aquatic plant management increases the competition for nutrients.
- Chemical control via allelochemicals (chemically produced by plants to control filamentous algae or other plant species) or algicides (contact or systemic sprays).

While all these parameters are measurable in terms of established methods, they cannot be used to measure the relative efficiency of a filamentous algae mitigation strategy since such a permit would be defined in terms of the drivers (determinants) of filamentous algae, which is problematic because:

- Several combinations of drivers (determinants) can yield the same level of algal growth (i.e. concentration substitutability exists between drivers).
- The drivers (determinants) per se do not necessarily have a direct negative impact on the output variables (agricultural production).

A direct measure of filamentous algae load is required. Consequently, it was decided to design the permit in terms of the outcome (i.e. algae). This allowed universal measure of the problem without affecting the scope of mitigation strategies addressing the drivers. Last-mentioned is commonly presented by ***chlorophyll concentration in milligram per square metre***, where high chlorophyll levels are indicative of high algal biomass.

Benthic chlorophyll-a is measured by spectrophotometry (absorbance or fluorescence) using either the known optical properties of chlorophyll or high-performance liquid chromatography. Benthic chlorophyll-a, which is linked to filamentous algae through the dried filtered biomass, is measured as milligram per square metre, which could then be used as a basic unit of measurement for a pollution permit.

Determine the pollution cap and number of permits

A pollution cap (i.e. a 'safe' filamentous algae load for the river or sections of the river) is required to determine the number of permits for the initial allocation. In effect, it thereby determines the scope of the market. The process starts by identifying an area of reference for the cap. Here it becomes important to understand that the area of reference needs to be an area where the conditions for filamentous algae growth are similar and where filamentous algae mitigation strategies will have comparable impacts (i.e. the areas need to be similar in terms of biophysical conditions for filamentous algae growth and the self-cleaning capacity of the river).

The identification of homogenous sub-areas could be done according to the self-cleansing properties of the river system, which have been indexed via a filamentous algae sensitivity index (Oberholster et al., 2013) and could be used to distinguish and identify homogenous sub-areas in the river. These areas need to be mapped by establishing monitoring points at the beginning and end of each area. The surface area of each sub-area needs to be calculated and the drainage map for the area needs to be confirmed. A map of these sub-areas (presenting the monitoring points and surface areas and drainage) provides the point of departure for defining the algae pollution cap for each area and the total pollution cap for the river.

The following boundaries could be used to classify algae loads (Table 11):

Table 11: Load boundaries for algae (Oberholster et al., 2013)

Boundary	Median benthic Chlorophyll-a (mg/m^2)	Median benthic Chlorophyll-a (g/ha)
Natural (oligotrophic)	<1.7	<170
Good (mesotrophic)	1.7–21	170–210
Fair (eutrophic)	21–84	210–840
Poor (hypertrophic)	>84	>840

Since the case study area is considered as having phosphorous-sensitive rivers (i.e. a eutrophic state can easily develop into a hypertrophic state), a load boundary of 21 mg/m^2 is considered desirable for the river. However, a pollution permit requires an absolute value for filamentous algae per sub-area and not a concentration. This implies the need for extrapolation based on the assumption of homogeneity to derive an absolute weight for the load for the sub-area. Consequently, the concentration value of 21 mg/m^2 is extrapolated to the surface area or surface flow area of the sub-area to obtain the total permissible pollution (i.e. the pollution cap in weight) for the sub-area.

For example, if the relevant area of the sub-area is equal to 15 ha, the total permissible pollution for the area is 3150 g (21 mg multiplied by 150 000 m^2). The number of permits issued is therefore subject to a constraint of 3150 g and could (in theory) be any number as long as it remains subject to the constraint of 3150 g for the subsection (see Table 12).

Table 12: Examples of permits subject to a hypothetical algal load of 21 mg/m^2 for 15 ha area

Number of permits	Representative weight per permit per area	Total (gr)
15	210 g per ha	3150
150	21 g per 1000 m^2	3150
1 500	2.1 g per 100 m^2	3150
15 000	210 mg per 10 m^2	3150
150 000	21 mg per m^2	3150

Although any of these combinations satisfies the 3150 g constraint, the relative weight of the permit will need to be standardised to allow trade between sub-areas. The choice of representative weight forms part of the terms and conditions of the permit, and should be based on practical considerations. For now, it has been assumed that the permit is defined in terms of gram per hectare, i.e. 210 g/ha, implying 15 permits for the specific area.

The reserve price and market-clearing price

The following example aims to present and explain some of the intricacies associated with the initial price determination process for a pollution permit. We have estimated the initial value of the impact of filamentous algae pollution in the Dwars River to be R1888 per hectare (refer to Section 12.4 and Table 9).

This estimate could now be used as point of departure to determine a reserve price (initial price) for a pollution permit in the area. This is the price considered to be the minimum amount that will leave a polluter indifferent between his WTP for a permit or WTA the losses of the pollution. According to Table 11, the value of filamentous algae will vary between R111/g (i.e. R1888 for 17 g) and R2.25/g (i.e. R1888 divided by 840 g), with the price at the preferred state (i.e. 210 g/ha) equal to R8.99/g (i.e. R1888 divided by 210 g). Trade now becomes possible between willing buyers and willing sellers. An example will illustrate the process:

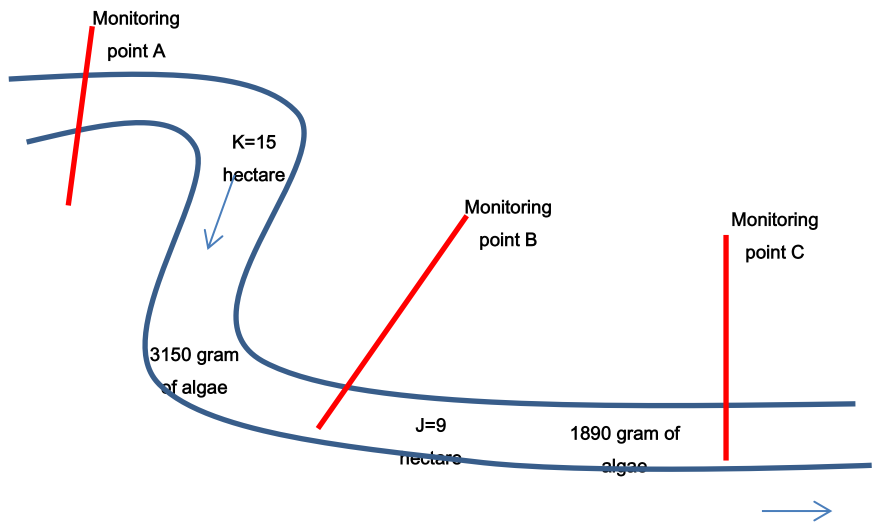

Figure 30: Hypothetical river section with sub-sections K and J

Consider Figure 30 with homogeneous sub-area K, which is 15 ha with a permissible amount of filamentous algae equal to 3150 g (15 ha multiplied by 210 g/ha). Sub-area J is 9 ha with 1890 g of filamentous algae, also at 210 g/ha. Direction of flow is from K to J.

Say at a given moment in time, the algae load at monitoring point B is 190 g/ha, (sub-area K is therefore doing well at 2850 g). Say the load increases downstream from point B and yields a reading of 220 g/ha at monitoring point C (i.e. an increase of 30 g/ha). This implies a total supply of algae of 1980 g (9 ha multiplied by 220 g) to sub-area J. This means that sub-area J has a WTP to get rid of 90 g (1980 g minus 1890 g) of algae or face a penalty/fine. Sub-area K has a current capacity of 300 g (3150 g minus 2850 g) and could therefore buy the 90 g and still remain within its limit if sub-area J can meet K's WTA price.

In order to calculate this price, one needs to understand that although the reserve price per hectare could be equal to the value of the impact of algae, i.e. R8.99/g (R1888 for 210 g), the WTA of sub-area K will start at R9.94/g (i.e. R1888 for 190 g) per hectare. Furthermore, the clearing price will be the new WTA of sub-area K once after assuming responsibility for an additional 90 g. This price will be close to R10.26/g, i.e. R1888 multiplied by 15 ha, which is then divided by 2760 g (i.e. 2850 g minus 90 g) per hectare. The value of the transaction is estimated to be R923.40 (i.e. R10.26 multiplied by 90 g) and the new target load for sub-area K will become 184 g/ha (i.e. 2760 g divided by 15 ha) at monitoring point B in order to meet the target load of 210 g/ha at monitoring point C.

However, say we keep the same scenario, but change the base value of sub-area K from 190 g/ha to 210 g/ha. The same 30 g/ha is now introduced downstream from point B with the reading at monitoring point C now becoming 240 g/ha. This implies that sub-area J needs to sell 270 g (i.e. 30 g multiplied by 9 ha) or face a penalty/fine. Sub-area K has no spare capacity and will have a WTA to R9.83/g, i.e. R1888 multiplied by 15 ha, which is then divided by 2880 g (i.e. 3150 g minus 270 g) per hectare. The value of the transaction will be R2654.10 (i.e. R9.83 multiplied by 270 g) and the new target load for sub-area K will become 192 g/ha (i.e. 2880 g divided by 15 ha).

Say we switch the two areas around, i.e. sub-area J is now upstream from sub-area K. If we keep the load at monitoring point B at 190 g/ha, and have the same increase of 30 g/ha realising a load of 220 g/ha at point C. Sub-area K will need to sell 150 g (i.e. 10 g/ha multiplied by 15 ha). Sub-area J's WTA will be equal to R10.89/g, i.e. R1888 multiplied by 9 ha, which is then divided by 1560 g (i.e. 1710 g minus 150 g). The value of the transaction will be R1633.50 (i.e. R10.89 multiplied by 150 g) and the new target load for sub-area J will be 173 g/ha (i.e. 1560 g for 9 ha). If we change the base value of sub-area J from 190 g/ha to 210 g/ha in this scenario, and introduce the same increase in load (i.e. 30 g/ha), then monitoring point C will read 240 g/ha. Sub-area K now needs to sell 450 g (i.e. 30 g multiplied by 15 ha). Sub-area J's WTA will increase to R11.80 (i.e. R1888 multiplied by 9 ha), which is then divided by 1440 g (i.e. 1890 g minus 450 g). The value of the transaction will be R5310.00 (i.e. R11.89 multiplied by 450 g) and the new target load for sub-area J will be 160 g/ha (i.e. 1440 g for 9 ha).

The above-mentioned are only examples of hypothetical cases and the process is subject to terms, conditions and transactional protocols.

13.2.2 Terms, conditions and transactional protocol

One could argue that a market is essentially a social construct facilitating the exchange of utility, which is regulated by accepted terms and conditions. Terms determine the transactional protocol (i.e. the trade process) while conditions specify the requirements of trade (i.e. requirements to participate in the market).

The terms and conditions should be designed within the context of existing regulatory regimes, i.e. in this case, water regulatory regimes of the area (typically a water user association or irrigation board responsible for the area). Terms and conditions also need to deal with complexities associated with ecological infrastructure and the associated ecosystem services, such as the public good nature of these services.

A detailed account of standard terms and conditions for tradable permit systems aimed at ecological infrastructure is presented in OECD (2001). However, the following aspects will need to be accounted for when designing the terms and conditions for the transactional protocol for water pollution permits:

- The non-point characteristic of pollution will need to be accounted for by homogenous (in terms of self-cleaning capacity) sub-drainage areas, which will need to be mapped and differentiated by means of monitoring points. This is an important cartographical exercise that underlies the permit system. Responsibility and accountability could be proportional to ownership of transformed land within the drainage area of the sub-area. If, for example, two owners own an equal amount of land within a particular sub-drainage area, the associated payment for – and income from – pollution permits (which is administered by the relevant water user association) will be covered and received in equal shares. Last-mentioned points towards the need that this water pollution permit system operates on co-operative principles, i.e. due to the non-point nature of pollution and the self-cleaning properties of rivers, sub-areas becomes the entities of trade, i.e. sub-areas trade with one another, not individual land owners.

- The frequency of monitoring algae loads at the monitoring points of sub-areas will be a necessary but not sufficient condition for determining the frequency of market activity since it is only after a monitoring event that the required data becomes available to identify problem areas and the need for transactions. Market activity (i.e. the need for transactions) will therefore be directly related to the frequency of monitoring events. The higher the frequency (i.e. the shorter the interval between monitoring events), the better the understanding of algae concentrations over time becomes, but the smaller the marginal difference between readings; hence, a lower number of permits and higher transaction costs. The lifespan of a pollution permit could be a function of the frequency of monitoring events (if these happen at regular intervals) or a timeframe negotiated with the members prior to the launch of the system. This lifespan will need to account for the trade-off between the self-cleansing capacity of the river and a fair amount of time to implement filamentous algae mitigation measures once a permit has been sold.
- A suitable incentive will need to be in place to incentivise parties in the wrong to engage in the market. Such an incentive could be a fine equal to the current market-clearing price plus a suitable premium in the form of a fine, which is big enough to incentivise market engagement. However, the parties in the wrong might have to pay the fine if a market solution (i.e. a transaction) cannot be found (i.e. no willing buyer/seller).
- A central regulative authority who verifies, facilitates (i.e. identifies and brings together potential buyers and sellers), validates, registers and records all transactions will need to be established. This function can reside with the water user association who then in effect hosts a ‘trade pool’ for transactions. This trade pool consists of ‘surplus’ pollution rights. It should increase with time if the river system health improves or if polluters opt to rather pay a fine instead of using the market to offset their pollution. The trade pool provides flexibility to the system since new entrants and polluters not able to find willing buyers could rent these permits from the trade pool. This would be subject to conditions such as they should provide an implementation plan to decrease their pollution within a given time period.
- The regulative authority is the only recognised authority in terms of facilitating the transactions. This will ensure that the same terms and conditions are used for all transactions.
- Consensus needs to be reached regarding a pollution cap (e.g. 210 mg/m^2 as per the example above).
- The terms and conditions should, once developed, be challenged in terms of compatibility with existing laws, regulatory frameworks and institutions by means of a pilot phase.

In order to guarantee security for tradable rights and therefore willingness to trade, it is not advisable to revise the terms and conditions of the game unexpectedly since it might jeopardise current investments in the scheme by affecting the value of the permits negatively. The conditions under which the rules of the game may be changed should be made explicit prior to launching the scheme.

13.2.3 The monitoring system

The monitoring process should be designed keeping prevailing regulatory structures (e.g. a water user association) in mind. It is understood that the measuring protocol for filamentous algae is a laboratory-based method that cannot currently be automated. The monitoring of the relative performance of filamentous algae mitigation strategies and the consequent information required to calculate the availability of permits would be a function of the frequency of monitoring. This is a management decision that will need to be negotiated before the scheme is launched. It is important to maintain the frequency of monitoring as it will affect price expectations and market stability. Once it has been determined, this timeframe should not be changed without good reason.

The result of each monitoring run presents important market information that needs to be verified and registered in the trade pool before being communicated to members.

Transactional monitoring continues with the administration and regulative processes of a transaction. Thus far, it has been suggested that the relevant water user association could establish a sub-directorate responsible for verifying, facilitating (i.e. identify and bring together potential buyers and sellers), validating and recording transactions in a register.

The verification and registration of intent (i.e. willing buyers of permits) will need to happen soon after the results of the monitoring run have become available. Verification is required to confirm that a potential seller, who registers intent, indeed has a right to sell. This is done by comparing the reading of the relevant monitoring point with the reading of the previous monitoring run and checking whether any credits are registered against this monitoring point in the trade pool or if it was traded to another owner. The transaction may then continue. Details of prospective buyers and sellers need to be recorded along with information such as physical location of the buyer and seller, the relevant monitoring points, and the amount of filamentous algae in question.

The facilitation process could be done in person between the buyer and seller, under supervision of the association, or via a lawyer. This is the step where price negotiation also takes place, which could be an automated process (i.e. an exchange) if the number of transactions justifies such a tool. The facilitation process aims to make it as easy as possible for willing sellers to find willing buyers (i.e. minimise transaction cost).

The validation process is then initiated (i.e. a bank transfer, or a reconciliation of the water user association accounts of the buyer and seller) once a clearing price has been determined. The transaction is recorded in a register and is then complete. No post-transactional monitoring is required since such monitoring is done with the next monitoring round. Clear and simple communication is pivotal for effective monitoring. This of course comes back to the need for a trustworthy relationship between the regulating system and its participants (i.e. farmers). Breetz et al., 2005 discuss this issue at length.

13.2.4 Pilot phase

A pilot phase will challenge the suggested terms and conditions of a water pollution trading scheme, and capture the many intricacies and additional special conditions required for implementation. This information should then be used as the basis for designing the legal framework for the scheme.

As the idea of buying and selling the ‘right’ to pollute is counterintuitive with potential conflicting opinions, a pilot phase will also provide opportunity for consultation and exchange with stakeholders and potential participants. This could improve the acceptance and buy-in and, therefore, efficiency of the formal legally established scheme. Here it is important to account for the fact that implementation should be done in consideration of the political acceptability of the system and risks associated with misinterpretations regarding the ‘right to pollute’ as well as the broader economic impacts such as inflationary impacts and market power balances.

13.2.5 Implementation via initial allocation

The sealed bid auction is a generally accepted method of allocating permits that is currently used. Buyers of permits must send their bids in a sealed envelope to the agency conducting the auction. The permits are sold to the highest bidders until there are no more bidders or the permits run out. There are two main features of sealed bid auctions that make them different from other methods. Firstly, they can be organised to prevent firms controlling a large fraction of the permits from exhibiting monopoly power. Secondly, they enhance price stability, which adds rational planning of pollution control by the polluters. Silent auctions are most efficient when permits can be traded freely at any time.

Alternatively, the initial issuing of permits could be done free of charge, but by means of application. This means that firms apply for permits to the regulation authority, who then registers and allocates permits subject to the overarching load per sub-area. Initial allocations should then be issued. Price determination of subsequent transactions is then a function of the WTP and acceptance of market participants as explained above.

14 DISCUSSION AND RECOMMENDATIONS

14.1 Discussion

It is evident from the findings presented in this study that CLDs have the potential to facilitate an alternative VCA where traditional approaches to VCAs are unsuitable. Despite not being able to conduct a VCA in the traditional manner, the outcomes of the scenario analyses allowed for an alternative VCA to be completed that achieved the relevant goals of traditional VCAs.

The ESVCA framework as developed in this study identifies forward linkages and ripple effects in individual value chains of final AESs. It also identifies and assesses challenges and opportunities in the value chains of final AESs and associated markets. It also provides a framework through which progress towards understanding and integrating fully inclusive VCAs, which incorporate environmental processes and services into policy and decision-making, are a realistic outcome. The model is predictive in nature and thus allows for a proactive approach towards ecosystem management geared towards improving the provision of chosen AESs while increasing system understanding for relevant stakeholders and decision makers. Private and public entities who rely on the provision of ecosystem services can take advantage of this approach to recognise potential opportunities and threats within the value chains of these ecosystem services. Identifying the most efficient methods of improving ecosystem service 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 corroborate the reliability of the subsequent VCAs. 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.

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 socioecological systems while simultaneously assessing assumptions of policy and practice aimed at improving human well-being through improved provision of essential ecosystem services. This is done by using the ecosystem service, CLD and value chain concepts to provide information to support long-term sustainable management of the socioecological systems upon which all life depends.

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 illustrated 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. Thus, effectively addressing all four research objectives of the study.

The approach could be used to inform the SEEA method regarding the links between anthropogenic activity and ecosystem services. 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 ecosystem services. Overall, it will provide a holistic system view that can be related directly to each service.

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 are the future of ecosystem service assessment, analysis and modelling. Combining complex models with 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 policymakers and management decision makers receive the most accurate, reliable and relevant information.

Ferraro (2000) argues that popular ecosystem development initiatives that attempt to increase the provision of desired ecosystem services are hindered in many ways and that direct contracting initiatives would be a more effective solution. Such contracting approaches create artificial markets through which individuals who provide ecosystem services can benefit. A worthwhile investigation would be to compare current conservation contracting programmes with the ESVCA framework and the learning associated with the market-making process for ecosystem service presented here in terms of their potential to conserve and promote ecosystem services in developing nations. This could potentially lead to the development of a combined approach that uses the framework and the market-making process to improve ecosystem functioning and further development of markets for ecosystem services.

Taking AES to market as a way of incentivising increased private sector investment to improve the service seems possible in theory. It could be done in several ways depending on the service in question. This focused on the market-making process of water pollution permits. The permit is considered the key of this strategy (which is only one of three strategies – refer back to Section 11.2 for other possible avenues for the private sector).

Although a tradable permit system is, due to its nature, a complex instrument in terms of implementation, it has immense potential to effectively mitigate pollution once up and running. This is mainly because polluters differ in their ability to abate their pollution – some can do it easily and cheaply, while it would be more difficult and costly for others. The freedom to trade pollution entitlements gives an incentive for polluters to consider abatement (since they can sell their surplus quotas) while others face the cost of having to purchase permits. For society, the existence of tradable permits enables pollution abatement to be achieved in a cost-effective way. Over time, pollution standards can be tightened, thus increasing the value of the permits and the pressure on market participants to pollute less.

This report presented some of the fundamental conditions and requirements for a water pollution permit system along with some of the processes that will facilitate implementation for South African river systems and their associated AES. Due to the novelty of the work, the focus was on the preliminary design of the system, a process which is ongoing. Consequently, no information was put forward regarding the regulatory environment (and the associated political process) that is required to facilitate acceptance among those affected. Last-mentioned should be investigated in a dedicated study. However, it should be clear that to have a functional market for water pollution permits, enough market participants (i.e. polluters) should exist (i.e. enough pollution should be present), which means that there should be many pollution sources (polluters) affecting the same parameter (e.g. nitrogen, phosphorous, biochemical oxygen demand, salinity) within the same catchment, while having significant differences in abatement cost curves so that beneficial trade becomes possible. Furthermore, the establishment of a legal entity (scheme administrator) responsible for monitoring and data capturing, enforcement mechanisms, information for conflict resolution, trade facilitation and system evaluation, will along with a solid scientific understanding of the pollution factors system responses, provide a solid basis for trade. However, care should be taken that the scheme administrator keeps administrative requirements, approval procedures and requirements to trade to a minimum since high transaction cost reduces willingness to trade.

It is also advisable that the existing functioning system of water pollution control should be in place to design and implement a market for tradable pollution permits. Such a system not only provides the basic context to design the permit system (ensuring compatibility), but it also streamlines the proposed system with existing data sources.

It should be noted that the pollution permit system is not a failsafe way to decrease the absolute level of pollution within a river and safeguard the supporting AES, but should rather form part of an integrated strategy and in conjunction with command-and-control regulations to mitigate pollution problems. In this case, the permit system has a particular role to avoid eutrophic conditions via relocating the pollution problem to sub-areas within the river that can best cope with the load at a particular time period.

14.2 Recommendations

It is evident from the broad array of objectives, concepts and methods adopted during this study that there are numerous potential future research endeavours that could generate original and relevant content to improve 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 to corroborate the findings of this study and address some of the fundamental limitations as far as possible.

Further research could also examine the commonalities between the specific outcomes of the approach presented in this study and those of ecosystem service accounting. The European Commission (2013) identified three key areas of research, two of which could be assisted using the ESVCA framework, namely, physical ecosystem accounting and communication and dissemination. Firstly, exploring the potential of ESVCAs to guide physical ecosystem accounting in terms of differentiating between final and intermediate ecosystem services for a specific beneficiary group 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.

The integrative framework designed for context-based decision-making in natural resources management presented by Pollard et al. (2014) 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. By incorporating the detailed ESVCA methodology into this framework, it could extend the basic scenario development approach to facilitate a more effective resilience analysis by using ecosystem service value chains. This should be investigated.

In terms of the market-making process,[5] these issues deserve further investigation, especially if it is clear that a minimum amount of pollution needs to be present before a market will function. This of course presents a fundamental contradiction to some: the market will not exist if no pollution is present, and if one creates dependencies on a functional market, one also creates incentives to maintain pollution.

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[5] A word of caution: along with the research on the market-making process, an effort needs to be made to assess the consequences (including livelihood impacts) of the privatisation of public goods such as AESs (see Corbera et al. (2007) and Corbera (2015)).

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APPENDIX 1: DESCRIPTIONS AND CLASSIFICATIONS OF THE VARIABLES AND ECOSYSTEM SERVICES

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 yield and quality of the water.	Section: Provisioning Division: Nutrition Group: Water Class: Surface/Ground water for drinking Class Type: *By amount, type*	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: *By reduction in risk, area protected*	Catchment and downstream residents.

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 *et al.,* 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 (-)

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 (+)

APPENDIX 2: SUMMARY OF INTERNATIONAL EXPERIENCE

International experience where the private sector engages in ecosystem services markets (references in this table not cited).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Muñoz-Piña, et al.	Democratic Republic of Congo	2011	Carbon.	US$240 040 (2009-2017).	Implemented through a local implementing agent that acted as facilitating agent; 200 seasonal and 30 full-time jobs.	Financial support of the Belgian Development Cooperation; sales of carbon to World Bank (via BioCarbon Fund) and private companies (Danone).	Private sector international companies.
Asquith, et al.	Bolivia	2003 onwards	Water.	PES start-up (~US$40 000) and running transaction costs (~US$3000 per year over the last three years).	Negotiated payment mode is annual *quid pro quo* in kind compensations in return for forest protection; facilitated by a local non-governmental organisation (NGO), Fundación Natura Bolivia.	One service buyer is an international conservation donor (the US Fish and Wildlife Service). The Los Negros municipal government has on their behalf contributed ~US$4500 to the scheme/external donors. The second service users are downstream irrigators who likely benefit from stabilised dry-season water flows if upstream cloud forests are successfully protected.	Downstream farmers.

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Clements, et al.	Cambodia	2010	Wildlife protection/ ecotourism (birdwatchers).	Approximately $120–$160 per family participating, and an average of $1200 or a maximum of $4000–$6000 per village.	The Ministry of Environment and Ministry of Agriculture, Forestry and Fisheries, with the support of the Wildlife Conservation Society (WCS), an international NGO, instituted a series of pilot PES programmes as a complement to protected area management in 2002. Project being managed by elected village committees/ WCS.	Fees paid by tourists/selling of goods/government.	Yes; tourists.
Gong, et al.	China	2006–2035	Multiple objectives of sequestering carbon, enhancing biodiversity, reducing soil erosion, and improving local livelihoods.	Expected total revenue from sales of carbon credits, timber and pine resin is approximately US$5.5 million: US$3.5 million from sales of timber and pine resin; and US$2.0 million from sales of certified carbon credits; government subsidies.	Paid as salaries.	Subsidised loans and government funding.	Yes (local forest companies).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Pagiola, S.	Costa Rica	2006	Water/ biodiversity conservation/ carbon sequestration.	About US$5 million (water services)/ US$5 million (biodiversity).	FONAFIFO (established eight regional offices to handle applications, contracts and monitoring). Landowners request first payment at contract signing, then subsequent payments made after verification of compliance.	3.5% of revenue of fossil fuel sales tax/World Bank loan/grant from Global Environment Facility (GEF)/German aid.	Yes (provision of water services; contribute to FONAFIFO administration costs).

Authors	**Country**	**Year**	**Services traded (e.g. carbon, water)**	**Monetary value**	**Mechanism (how transaction took place)**	**Incentive (what money and where does it come from)**	**Private sector involvement**
Blackman, et al.	Costa Rica	2010	Biodiversity, carbon sequestration, scenic beauty, and hydrological benefits.	Annual/ha payments were US$40 in 1997, when the PSA programme was created, rose to US$43 in 2005, and to US$64 in 2006.	The forest protection modality requires landowners to preserve primary or secondary forest cover on their land for five years, a commitment that can be renewed. FONAFIFO makes a partial payment to the landowner when they sign a programme contract. Subsequent payments are made only after a third party verifies that tree cover has not been cleared.	FONAFIFO's funding for payments to land managers has been derived from four sources; i) Tax revenue – from a national tax on gasoline (3.5% of the total gasoline tax); ii) supplemented since 2006 by revenue from a national tariff on water use (25% of the total tariff); this is the main continuing source of funds for the programme; iii) loans (to be repaid with tax revenues) and grants from the International Bank for construction and development and grants from the GEF have financed payments for 45% of all hectares enrolled in FONAFIFO programmes/The German International Development Bank (Kreditanstalt für Wiederaufbau) has financed another 10%. Finally, a variety of users have financed FONAFIFO payments to 3% of total hectares.	Yes, hydroelectric sector (as part of user financing).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Corbera, et al.	i) Guatemala	2007	Biodiversity conservation; Watershed conservation; Carbon dioxide fixation/activities promoted include protecting and managing the hydrographical basin; encouraging sustainable agricultural practices; providing opportunities for low impact ecotourism; and promoting sustainable forest management through agroforestry.	Payment of US$17.86/ha/year.	The Fundacio´n para el Ecodesarrollo y la Conservacio´n (FUNDAECO), which administers the Reserve on behalf of the Guatemalan State, negotiated a PES scheme with the Empresa Hidroele´ctrica del Atla´ntico (HEDASA), a local hydroelectricity company, on the premise that an increase in forest conservation efforts would ensure continuous water flows and a reduction in sediment loads.	PES funds come from an increase in the water tariff of US$0.20/month.	Yes, a local hydroelectricity company).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Corbera, et al.	ii) Nicaragua	2007	Biodiversity conservation; watershed conservation; carbon dioxide fixation; landowners commit to avoid fires before, during and after sowing; develop organic agriculture; conduct soil conservation practices; develop agroforestry systems; promote fore regeneration and commit to prevent livestock from invading the PES areas.	Each household contributes with US$0.31/month to the PES scheme and landowners receive US$26/ha/year. Landowners commit to avoid fires before, during and after sowing; develop organic agriculture; conduct soil conservation practices; develop agroforestry systems; promote fore regeneration and commit to prevent livestock from invading the PES areas.	Problems regarding water quality and quantity led 125 households from San Pedro del Norte to propose and negotiate a PES scheme with the support of a local NGO (PASOLAC) and other regional civil organisations, which identified priority areas for funding in the upstream basin recharge area. The 125 households created a water committee and reached five individual agreements with upstream landowners, covering a total of 39.2 ha for reforestation and conservation of the prioritised areas.	The 125 households created a water committee; each household contributes with US$0.31/month to the PES scheme.	Yes (negotiate a PES scheme with the support of a local NGO (PASOLAC) and other regional civil organisations).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Corbera, et al.	iii) Mexico	2007	Biodiversity conservation; watershed conservation; carbon dioxide fixation.	The project secured funding for the sale of 60 498 t of carbon dioxide equivalent (tCO2eq)2 over 30 years at a price of US$3.27/tCO2eq, from which 66.6% (US$2.18/tCO2eq) is allocated directly to farmers, and the rest is used to cover project administration and managers' salaries.	Participant farmers and communities rely on subsistence and semi-subsistence maize and bean cultivation, livestock and relatively little commercial agriculture. The project's objective is to provide carbon benefits through forestry systems, which are economically viable, and socially and environmentally responsible. There is 4738 ha under reforestation and conservation activities funded by several investors: The Carbon Neutral Company, TetraPak, International Automobile Federation and The World Bank, which in exchange receive VERs to offset their greenhouse gas emissions and to provide carbon neutral products and services to their clients.	The project's objective is to provide carbon benefits through forestry systems, which are economically viable, and socially and environmentally responsible. There is 4738 ha under reforestation and conservation activities funded by several investors: The Carbon Neutral Company, TetraPak, International Automobile Federation and The World Bank.	Yes (reforestation and conservation activities funded by several investors): The Carbon Neutral Company, TetraPak, International Automobile Federation and The World Bank.

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Corbera, et al.	iv) Belize	2007	Biodiversity conservation; watershed conservation; carbon dioxide fixation.	The project expects to sequester 10 million tCO2eq over the period 1995–2035 with a total expenditure of US$2.6 million in the first 10 years and US$3 million in the following 30 years. This translates into an approximate undiscounted price of US$0.25/tCO2eq.	The project has involved one international conservationist organisation, The Nature Conservancy (TNC), and one consultancy firm, Winrock International, in brokering an agreement with investors, and preparing carbon sequestration scenarios and forest management plans, respectively.	Investors include a consortium of US and Canadian energy utilities.	Yes (The project has involved TNC and Winrock International. Investors include a consortium of US and Canadian energy utilities).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Turpie, et al.	South Africa	2008	Water.	The WfW programme has an annual budget of more than R2 billion; a fraction of which comes from the water trading account. Water trading account is private sector money earmarked for catchment management.	WfW is a public agency under the jurisdiction of the Department of Water Affairs and Forestry (DWAF) with the mandate of controlling invasive alien plant infestation; job creation; salaries.	The bulk of the funding over the last 11 years has been generated through poverty relief programmes (the Reconstruction and Development Programme, then the Special Public Works Programmes, which evolved to become the Expanded Public Works Programme); WfW effectively acts as a conduit for the provision of ecosystem services, predominately water supply, through the control of invasive alien plants and the provision of unskilled job opportunities, using predominantly taxpayers' money. The DWAF includes a water resource management fee in the water tariff charged to consumers.	Yes; through the trading account.

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Montalvo, et al.	Mexico	2011	The Scolel' Te programme is a community carbon management scheme. Carbon service generating activities are afforestation, reforestation, agroforestry, forest conservation and restoration.	Payments made in 2010 total US$109 584.91. These payments covered pending payments from previous years and those corresponding to 2010.	Payments to communities by programme after evaluation.	Payments from carbon sales.	Private sector buyers of carbon credits.

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
McElwee	Vietnam	2012	Landscape protection, ecotourism, carbon sequestration.	Lam Dong Province decided that of the 2009 fees collected (47 billion VND; US$2.61 million), 10% will be kept in the provincial fund to cover expenses, 9% of the fund will go to 13 large forest owners (such as SFEs and FPMBs) to cover their costs, and 81% will go to household payments. For a household, this will amount to around 280 000 VND (US$15)/ha in payment, with participating households having between 10 ha and 30 ha on average to protect.	Not specified. Forest service suppliers in the pilot have included households and individuals near forest lands who have contracts or red books via FPMBs and SFEs operating in the area. These participating forest goods suppliers must enter into a contract with the local government agreeing to forest protection conditions before they are entered into the PES contract.	Various potential actors (government, conservancies and private).	Yes.

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Branca, et al.	Tanzania	2009	Watershed services.	It is estimated that the reduction in sediment load in Ruvu River, resulting from the implementation of sustainable land management (SLM) practices in the Ulugurus area, could reduce Dar es Salaam Water Supply and Sewerage Corporate (DAWASCO) treatment costs by 10% (i.e. 200 000 US$/year). It has been also estimated that by 2018 DAWASCO could reduce total costs – both reduced costs and the saving of costs, which would otherwise be incurred – by more than US$400 000/year.	The pilot phase of the Equitable Payments for Watershed Services (EPWS) programme will involve 1215 households who will receive support in changing the current agricultural practices and implement SLM interventions over the total farmland area of 2240 ha (CARE/WWF 2007c). Incentives are to be in the form of in kind payments (vouchers that participants are free to use as they wish), provided to the farmers who have already implemented part of the practices (as a payment for work done), and calibrated to compensate the costs associated with implementation and maintenance of SLM practices. The contractual framework under which SLM practices are adopted and in kind payments made, involves the aggregation of land owners and disbursement of in kind payments by village authorities.	CARE/WWF identified two buyers that showed the highest willingness and ability to pay for reduced water treatment costs as a result of SLM practices implementation: the public water utility DAWASCO and the private company Coca-Cola KL (Kwanza Limited) to which DAWASCO supplies water. DAWASCO has agreed to contribute US$100 000 over 4 years to the EPWS programme. Coca-Cola KL has agreed to contribute US$200 000 over the same period as an initial form of payment that should help farmers to overcome the costs to the adoption of the SLM measures (CARE/WWF, 2007b, e).	Yes (as buyer of services/funding through in kind vouchers).

Authors	Country	Year	Services traded (e.g. carbon, water)	Monetary value	Mechanism (how transaction took place)	Incentive (what money and where does it come from)	Private sector involvement
Frost et, al.	Zimbabwe	2008	Wildlife protection/ ecotourism.	Zimbabwe's Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) generated over US$20 million of transfers to the participating communities, 89% of which came from sport hunting.	CAMPFIRE, a community based natural resource management programme in which rural district councils, on behalf of communities on communal land, are granted the authority to market access to wildlife in their district to safari operators. These in turn sell hunting and photographic safaris to mostly foreign sport hunters and eco-tourists.	The district councils pay the communities a dividend according to an agreed formula. In practice, there have been some underpayments and frequent delays.	Yes, hunters.