Towards the Development of Economic Policy Instruments for Sustainable Management of Water Resources

Report to the
Water Research Commission

by

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EXECUTIVE SUMMARY

Economic development in South Africa results in increased pressure on aquatic ecosystems. In order to mitigate the resultant degradation, various government departments have policies in place to regulate activities that damage ecosystems. These policies broadly follow a regulatory approach, adopting a mitigation hierarchy which seeks to avoid or minimize degradation impacts. Where impacts cannot be avoided, this regulatory approach prescribes compensatory mitigation, in the form of rehabilitation or restoration activities, to offset these impacts.

The policy instruments that govern these mitigatory activities are termed regulatory instruments. They principally comprise environmental impact assessments and their associated regulations; as well as water use licenses and their associated regulations. These regulatory instruments take a threshold approach and thus seek to set limits to environmental degradation.

To be successful at a water basin-wide scale, the regulatory threshold approach requires significant State financial and human resources to enforce compliance. This includes monitoring, evaluation, policing and punitive treatment of offenders. It also requires an adaptive management approach where decision-makers and managers are able to learn by doing, thus enabling decisions to be taken based on emerging knowledge which can be checked and refined over time.

It is common cause that the South African fiscus is under great pressure, and thus, the required financial resources are insufficient.

Thus, while we are striving towards a mature regulatory system (as above), we need to be more innovative in finding additional policy instruments that could curb environmental degradation.

Economic policy instruments (EPIs) are such an innovative approach.

A policy instrument is the term used to describe the methods used by governments to achieve a desired effect as envisaged in policy. Three types of environmental management policy instruments exist: regulatory instruments, suasion instruments and economic instruments. Regulatory instruments are by far the most commonly used environmental policy instruments internationally. Examples include laws of a rationing or prescriptive nature; and regulations that permit or license resource use, planning controls or performance standards. Suasion instruments are ethical or discretionary instruments that use moral and direct persuasion to promote appropriate behaviour. Economic instruments, which are the focus of this study, seek to influence behaviour and decision-making through introducing economic incentives or disincentives. Their purpose is two-fold: to achieve policy objectives and to earn revenue, the so-called double-dividend.

It is to be noted that EPIs are not viewed as an alternative to regulatory and suasion instruments. Rather, in the design policy of EPIs, it remains important to combine economic policy instruments with appropriate elements of regulatory and suasion instruments.

Within the above context, the aims of this assignment were three-fold:

- Conceptualise and describe the use of policy instruments both at an international and national level with special focus on economic policy instruments as a mechanism for reducing environmental degradation (focussing on the water sector).
- Investigate and define the scope of limitations faced by current EPI's in a South African context.
- Explore and demonstrate possible alternative EPIs, based on limitations identified, that would provide suitable mechanisms for internalising environmental damage into the economy.

We developed a conceptual framework for EPIs for the water sector through a triangulation of critical literature review, ecosystem services analysis and South African water policy review. In literature, EPIs have been referred to as market-based instruments (MBIs) or economic instruments (EIs).

Although literature on the subject of EPIs pre-date the 2001 World summit, the conceptualisation of "ecosystem services" through the Millennium Ecosystems Assessment (MEA) process, helped to expand research in this field. In 2004, UN Environment (previously "UNEP") published a discussion document on EPIs as an alternative to pure regulatory instruments. This document drew largely on concepts put forward by a World Bank study (1998). Analyses of these reports and their supporting literature reveals that EPIs programmes predominantly are designed as fiscal instruments that seek to raise revenue, and that environmental policy goals were largely not achieved.

Thus, effective EPIs should seek a double-dividend goal. The International Monetary Fund (IMF), in 1998, investigated a concept which it termed *"environmentally motivated fiscal policies"* that would yield *"win-win' outcomes in the sense that they improve environmental quality as well as attain macroeconomic objectives, such as more employment or a higher rate of economic growth"*. Some authors also refer to the concept of a "double-dividend", thus benefitting both environmental quality and natural capital stocks on the one hand, and employment and GDP growth on the other hand.

Central to achieving a double-dividend outcome, is the Nash Equilibrium. In a Nash Equilibrium, role players in a market can each achieve optimal outcomes simultaneously. All role players win, because all get the outcome they desire. Three sets of role players exist in our case:

- Government acting in the public interest on behalf of public goods, in this case water resources and water,
- Water resource users and water users, and
- Beneficiaries of aquatic ecosystem services.

The concept of ecosystem services is very helpful in developing a conceptual EPI framework. The MEA and TEEB define ecosystem services as the direct and indirect contributions of ecosystems to human well-being. They distinguish between four types of ecosystem services: provisioning, cultural, regulating and supporting services. This framework enables the analysis of chains of causality linking ecosystems to market role players and provide fundamental insights into EPIs.

EPIs as market-based instruments need Government (regulation entities) to play an oversight role to ensure that these markets operate efficiently. Finding the right regulatory framework and the right amount of regulation requires a delicate balancing act. Under-regulation may allow for fraud and manipulation while over-regulation may be too costly to comply with. Market oversight mechanisms may comprise a range of institutional arrangements (policies, protocols, systems, organisations and trading platforms) that we collectively refer to as transaction clearing mechanisms. In finance, "clearing" denotes all activities from the time a commitment in a transaction is made, to the point where the transaction is settled. Mature markets clear transactions through a third party, known as a clearing house. Designing for effective EPIs, therefore, need to design for effective clearing houses.

This report is structured into five chapters. Chapter 1 provides an introduction to the project. Chapter 2 develops the definition for EPIs through a literature review. Chapter 3 investigates the barriers to implementing EPIs in South Africa, based on policy review and expert interviews. Chapter 4 proposes an EPI framework for the water sector in South Africa, and identifies seven types of EPIs, relevant to water management:

- A basket of policy relevant (equity, efficiency, sustainability) water use charges
- Green infrastructure management systems
- Eco-restoration
- Waste discharge charges
- Industrial wastewater charges
- Pollution deposit-refund system
- Water pollution permit trading.

Chapter 5 further evaluates four of these EPIs, arbitrarily chosen, with the purpose of evaluating the conditions for feasibility of implementation.

The findings of the work illustrated that EPIs could internalise ecosystem benefits into the economy in such a way that it can achieve "double-dividend" (i.e. achieve policy imperatives and be profitable) outcomes.

Several conditions for successful EPI implementation exist:

- The policy objective(s) of the EPI need to be clearly defined
- EPIs should be complimented by suitable regulatory instruments
- The process is initiated by the regulator, i.e., in the case of water, DWS, DEA or WSAs
- Ecosystem service benefits need to be clear and measurable
- Users / impactors need to be clearly identified
- Benefits and beneficiaries need to be clearly identified

- An appropriate combination of transaction clearing mechanisms needs to be applied
- Transaction costs need to be controlled
- Finally, the private sector would play a key role in implementing EPIs and need to be involved from the outset.

Our key conclusions from this work are as follows:

- If the primary purpose of EPIs is to raise revenue, it would likely fail. Central to achieving a double-dividend outcome, is the concept of Nash Equilibrium, which envisages that role players in a market can each achieve optimal outcomes simultaneously. All role players win, because all get the outcome they desire.
- Implementation of EPIs requires both regulators and other implementers to be innovative and forward looking. The findings of the 2004 UN Environment indicated that "*lack of understanding of how EIs work; political interests that seek to maximize control costs via regulation; and a preference for keeping the status quo*" hampers effective EPI implementation.
- For EPIs to be effectively implemented, a range of suitable transaction clearing mechanisms, and institutional arrangements need to be in place. It is of little use to design sophisticated market-based systems if existing regulatory instruments and their institutional arrangements are not effective (refer to section 4.2).
- Effective EPIs require public awareness to be high and uncertainty low. These factors, when combined with a mistrust of regulators and/or firms responsible for environmental degradation, would lead to weak or obstructive participation among stakeholders.

The domain of EPIs, as defined in this report, allows for highly innovative conceptualisation of specific EPIs to address specific policy objectives. Thus, although the South African landscape of transaction clearing mechanisms and institutional arrangements may not be fully mature, it is possible to design EPIs that build on existing areas of strength.

We therefore propose, for further work, that the WRC identifies EPIs with high likelihood of near-term success and conduct in-depth case studies, at feasibility level of accuracy, on these, with a view to supporting the development of the National Water and Sanitation Master Plan. The case studies would have to:

- Set clear policy objectives and achieve these
- Develop, with detailed specifications, the transaction clearing mechanisms required to institutionalise the EPIs
- Perform feasibility testing, using appropriate regulatory and valuation assessment techniques.

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ABBREVIATIONS AND ACRONYMS

AMD	Acid Mine Drainage
CAC	Command and Control
CBA	Cost Benefit Analysis
CGE	Computable General Equilibrium
CMA	Catchment Management Agency
CMS	Catchment Management Strategy
COD	Chemical Oxygen Demand
COGTA	Department of Cooperative Governance and Traditional Affairs
DEA	Department of Environmental Affairs
DMR	Department of Mineral Resources
DORA	Division of Revenue Act
DWS[1]	Department of Water and Sanitation
EEA	European Environmental Agency
EEB	Economics of Ecosystems and Biodiversity
EI	Economic Instruments
EIA	Environmental Impact Assessment
EPI	Economic Policy Instrument
ES	Ecosystem Services
ETS	European Union's Trading System
EWR	Ecological Water Requirement
FAO	Food and Agriculture Organization of the United Nations
FBWP	Free Basic Water Policy
GDP	Gross Domestic Product
GWS	Government Water Schemes
IRMSA	Institute of Risk Management of South Africa
MARDV	Maximum Allowable Resource-Directed Values
MBI	Market-Based Instruments
MEA	Millennium Ecosystem Assessment
MIG	Municipal Infrastructure Grant
MPRDA	Mineral and Petroleum Resources Development Act
MWIG	Municipal Water Infrastructure Grant
NEMA	National Environmental Management Act
NWA	National Water Act
NWPR	National Water Policy Review
NWRI	National Water Resource Infrastructure

[1] Department of Water and Sanitation (DWS) was known as Department of Water Affairs (DWA) until 2014. Therefore, many publications still use DWA as their reference. We will, for simplicity's sake, consistently use DWS when referring to this department.

NWRS	National Water Resource Strategy
O&M	Operations and Maintenance
PES	Payment for Ecosystem Services
PPP	Polluters Pay Principle
REC	Recommended Ecological Category
RoA	Return on Assets
RQO	Resource Quality Objectives
RRDV	Recommended Resource-Directed Value
SFWS	Storage Framework for Water Services
SRP	Soluble Reactive Phosphorous
TEEB	The Economics of Ecosystems and Biodiversity
TIN	Total inorganic nitrogen
UN	United Nations
UNEP	United Nations Environmental Programme
USAID	United States Agency for International Development
USEPA	United States Environmental Protection Agency
WDCS	Waste Discharge Charge System
WfW	Working for Water
WMA	Water Management Area
WQ	Water Quality
WRC	Water Research Commission
WRM	Water Resource Management
WRMC	Water Resource Management Charge
WRMP	Water Resource Management Plan
WSA	Water Service Authority
WSP	Water Service Provider
WTE	Water Trading Entity
WUAs	Water User Associations
WUL	Water Use License
WWTW	Wastewater Treatment Work

GLOSSARY

Economic externality – These are costs borne by economic actors other than the polluter. In the case of water resources, the most common economic externalities are those associated with the impact of an upstream diversion of water, or pollution discharge, on downstream users. There are also externalities due to over-extraction from common pool resources such as underground water.

Economic Policy Instrument (EPI) – Policy instrument that attempt to influence behaviour and decision-making through introducing economic incentives or disincentives into the economic decision-making processes.

Ecosystem services – These are the benefits derived from ecosystems, also described as environmental goods and services. Ecosystem services in this report are as defined by the Millennium Ecosystem Assessment Framework of ecosystem services.

Environmental externality – These are costs borne by society as a result of ecological damage. Environmental externalities generally reduce human well-being through their impacts on ecosystem health and public health.

Fiscal cost – These are costs incurred by government, and particularly National Treasury. In this report, fiscal costs specifically relate to weaknesses in the water tariff system that reduces revenue collection from water use.

Payments for Ecosystem Services (PES)- refers to a voluntary transaction in which the generation of a clearly defined ecosystem service is being paid for by a buyer from a seller, only if the seller secures the provisioning of that service

Transaction Clearing Mechanisms- institutional arrangements required for clearing transactions within the EPIs. Clearing here is defined as in the financial market sense, i.e. all the activities required from the moment a commitment is made to the point of settlement.

Water price – A water price is a market price which is determined through negotiation between buyers and sellers.

Water resources – Water resources are defined in the National Water Act (NWA 1998) to comprise watercourses, surface water, estuaries and aquifers. Water resources are therefore natural assets that form part of the ecological infrastructure of South Africa.

Water tariff – A water tariff is an administered price assigned to a water resource management tariff, a water supply tariff or a sanitation tariff for services supplied by public entities to their customers.

Water tariff system – This is the system of institutional arrangements required to set water tariffs, send invoices and collect revenues

1 INTRODUCTION

The incorporation of the economics of aquatic ecosystems and biodiversity (EEB) in pricing of water, and its subsequent management, remains a global challenge but is especially important in South Africa within its increasingly water scarce context. According to the Institute of Risk Management South Africa (IRMSA) 2017 risk report, water related concerns have rapidly shifted to the top of the agenda for most organisations operating in the country. These concerns have resulted in ranking water crisis as the country's second biggest risk, just behind corruption and above economic slowdown and unemployment. This risk results from several hazards: urban migration puts pressure on cities to meet their water demands; water infrastructure investments have lagged behind rates of increasing water demand; and the most severe drought to have hit the country in decades. The mini budget speech on the 25th October 2017 by the Minister of Finance indicated that National Treasury is considering investing in water infrastructure, with the hope to release pressure on water demand and aging infrastructure.

Ecological degradation that exacerbates water crises is attributed to various land use activities and intensities broadly including, but not limited to, agriculture, industrial, residential and commercial. The impacts that varying land use intensities have had on aquatic ecosystems have been extensively investigated (Azrina et al*.,* 2006; Lenat and Crawford, 1994; Maldonadoo, 2010; Qu et al*.,* 2010; Winter and Duthie, 1998). Results show that with increasing intensity there is an increased pressure on these natural systems.

Cumulative ecological degradation which is caused by habitat loss; alteration of flow regime; point and no-point pollution results in risks that are borne by various economic sectors and humans directly, and especially poor communities who are reliant on aquatic ecosystem services for their livelihoods. These risks are often poorly understood and therefore also poorly managed.

The lack of understanding of regulating ecosystem services is considered the main reason for overexploitation and degradation of ecosystems assets and the ecosystem services they provide to humans (MEA, 2005; 2007; TEEB, 2010; WAVES, 2013). This has directly contributed to overexploitation and degradation of ecosystems, especially in developing countries, and has, arguably, also led to often extreme precautionary and risk-averse environmental policy in some countries (Sunstein, 2003; Pindyck, 2006).

In order to improve environmental risk assessment, the MEA (2005) and subsequently The Economics of Ecosystems and Biodiversity – TEEB (2010) introduced a new way of thinking about the value of indirect services of ecosystems (regulating and supporting) underlying the direct benefits derived by human society from ecosystems. The MEA and subsequently TEEB define ecosystem services as the direct and indirect contributions of ecosystems to human well-being distinguishing between four types of ecosystem services: provisioning, cultural, regulating and supporting services.

The regulating services are the services that ecosystems provide by acting as regulators, and are closely associated with the supporting services, which underpin almost all other services through the function of providing living spaces for humans, plants and animals. Regulating services thus play an indirect role in the economy and mitigate environmental risks.

Yet, in spite of our improved understanding and our improved valuation techniques, evidence of severe ecological degradation is evident all around us (UNEP 2004, MEA 2007). Several recent WRC studies have, in particular, highlighted the trend in water resource degradation (Maila et al., 2018; Maila et al., 2017).

South Africa has legislation and policies in place that regulate activities that damage ecosystems. These include environmental impact assessments (EIAs), water use licenses and broad environmental damage regulations. These instruments are not sufficient on their own to curb ecological degradation. There may be many reasons for this, and this project will try to understand their limitations.

Although some international examples of such incorporating EEB into water resource management exist, these are not easily transferrable, for three reasons. Firstly, South African river basins have unique socio-ecological settings comprising unique ecosystems, ecosystem services bundles and socio-economic conditions. Secondly, economic realities and water policy imperatives create unique supply and demand settings. Thirdly, the country is characterised by unique institutional settings.

This study therefore investigates potential economic policy instruments for internalizing ecological values into the South African water economy. The aims of this study are:

- To conceptualise economic policy instruments for water management as a clearing mechanism for mitigating aquatic ecosystem risks
- To demonstrate the functionality of the proposed economic policy instruments for water management by means of a suitable analysis framework using a case study approach.

2 LITERATURE REVIEW

2.1 Concept: Economic Policy Instruments

Environmental management “policy” is described by UN Environment (formerly “UNEP”) as any course of action deliberately taken, or not taken, to manage human activities with the view to prevent, reduce or mitigate harmful effects on nature and natural resources and ensuring that the anthropogenic changes to the water resources and surrounding environment do not have harmful effects on humans (UNEP, 2004).

“Policy instrument” is the term used to describe the methods used by governments to achieve a desired effect as envisaged in policy. Broadly speaking, three types of policy instruments exist: regulatory instruments, suasion instruments and economic instruments.

Regulatory instruments are by far the most commonly used policy instruments locally and internationally. Examples of these instruments include laws of a rationing or prescriptive nature; and regulations that permit or license resource use, planning controls or performance standards. A ‘Command and control’ (CAC) approach is mostly exercised in conjunction with laws and regulations. ‘Command’ refers to standards or targets set and that are to be complied with; and ‘Control’ refers to the enforcement of compliance. Regulations and standards generally desire to achieve a uniform level of control, but they can be inflexible. Regulatory instruments have traditionally been the key policy instrument type used in internalising ecosystem impact into the economy and will continue to play a key and likely increasing role in future

Suasion instruments are ethical or discretionary instruments that use moral and direct persuasion to promote appropriate behaviour (Roman, 1996). Voluntarism and corporate social responsibility are additional key suasion instruments. Education and information instruments are also very important key suasion instruments. When economic actors lack the necessary information about the environmental consequences of their actions, they may act inefficiently. The range of educational and information-based instruments is broad and can involve varying degrees of compulsion by government (Crafford et al., 2016).

Economic Policy Instruments (EPI), are the third type of policy instruments which, by definition, attempt to influence behaviour and decision-making through introducing economic incentives or disincentives into the economic decision-making processes. Typically, these instruments attempt to influence prices or costs to either increase or reduce demand for specific water benefits, with the purpose of incentivising certain desired micro-economic behaviour. These instruments are therefore used as a way of influencing the actions of individuals and corporations through monetary and fiscal instruments. By example, these may include subsidies, taxes and fees, tradable permits, administered tariffs, or production incentives.

EPI's therefore form the focus of further analysis in this assignment. It is to be noted that economic instruments are not viewed as an alternative to regulatory and suasion instruments. Rather, in the design policy of EPIs, it remains important to combine the EPIs with appropriate elements of regulatory and suasion instruments.

2.1.1 Classification of Economic Policy Instruments

EPIs are intended to induce some desired changes in the behaviour of all water users into the economy and to reach the environmental objectives at the least cost to society. EPIs offer instruments that encourage behavioural change through market signals in the form of a modification of relative costs, financial transfer, or both, ranging from tradable permits and pollution taxes, to deposit-refund systems and performance bonds (OECD, 1994; PCE, 2006, UNEP, 2004). It is also recognised that EPIs can complement regulatory instruments, thus increasing policy effectiveness and achieving environmental objectives at a lower cost (CEC, 2000; Jordan et al., 2003, UNEP, 2004).

EPIs have also been defined as a particular type of institution, a technical device with the generic purpose of carrying a concrete concept of the politics/society relationship and sustained by a concept of regulation (Lascoumes and Le Galès, 2007). This definition is narrow, as we will demonstrate that successful EPIs require multiple devices for implementation. Sterner (2003) does not put forward a concise definition for policy instruments but refers to various typologies for classifying such instruments.

Nicolaisen *et.al.* (1991) classified EPIs into five main categories (Table 2.1). An important attribute of the EPIs is the freedom of choice they leave where the impactor may choose the most beneficial solution themselves. Thus, a main reason to adopt EPIs is their potential to provide the same outcome as CAC, but through a lower financial cost to industries and with a lower net social cost.

Table 2-1: Classification of Economic Policy Instruments according to Nicolaisen and co-workers (1991)

Economic Instruments	Description
Pollution charges	A polluter pays the price of environmental pollution in the form of fees or taxes.
Market creation	Creating a market in which economic agents may buy or sell the "right" to cause pollution (tradable permits).
Subsidies	Subsidies include grants, tax incentives and low-interest loans, and serve to incentivise the impactor to change their behaviour and reduce pollution, or to lower the cost of pollution-reducing measures.
Deposit- refund systems	The consumer pays an extra charge (deposit) when buying a potentially polluting product, which is compensated when the product is returned to an approved recycling centre. Applied on products such as beverage containers and automobile batteries.
Enforcement incentives	Economic measures related to direct regulation, aimed to encourage dischargers to meet environmental standards and regulations (this type of instrument may be viewed as a regulatory support measure rather than a purely economic tool).

There is also consensus that a combination of policy instrument types is required to deal with complex policy objectives related to environmental issues (Bennear and Stavins, 2007; Helm, 2003; OECD, 2003b). These policy issues have particularly stressed the need to deal with complex issues in a comprehensive and broad way. For instance, by combining regulatory and EPIs, regulations may limit the production of a given product by requiring the producers to obtain permits for its production (and issuing only a limited number of such permits), while at the same time allowing trade in these permits between producers. Examples include tradable air pollution permits, or greenhouse gas allowances in the U.S. and the EU.

2.1.2 Market-Based Policy Instruments

Market-based instruments (MBIs), as defined by the World Bank (1991) and elaborated on by Sterner (2003), are a poorly explored and developed set of instruments in the water sector.

The World Bank (1998), defined MBIs as instruments that "*attempt to align private costs with social costs to reduce negative environmental externalities*". The study defines as MBIs all instruments that involve any form of monetary transaction. These include, on one extreme, a set of so-called "weak" MBIs, which includes penalties for not adhering to regulatory requirements, as well as different forms of taxes with the purpose of raising fiscal revenue. On the other extreme are so-called "strong" MBIs where market forces are harnessed to achieve environmental policy goals.

This World Bank Study concluded that EPIs programmes predominantly had been designed as "weak" MBIs (or fiscal instruments) that intended to raise tax revenue, and that environmental policy goals had largely not been achieved. The later (2004) UN Environment study concluded similar findings and identified a number of barriers to EPI implementation, principally a "*lack of understanding of how EIs work; political interests that seek to minimize control costs via regulation; and a preference for keeping the status quo*" (UNEP, 2004).

Rather, to be effective, EPIs require a "double-dividend" approach. This means that both environmental quality and natural capital stocks on the one hand, and employment and GDP growth on the other hand (including raising of revenue) need to benefit (Lighthart, 1998). The IMF (1998), also investigated a concept which it termed *"environmentally motivated fiscal policies"* that would yield *"win-win'' outcomes in the sense that they improve environmental quality as well as attain macro-economic objectives, such as more employment or a higher rate of economic growth"*.

However, the double-dividend effect has been explored predominantly in literature on environmental taxes. The hypothesis is that increased taxes on polluting activities would provide two kinds of benefits. The first was an improvement in the environment, and the second an improvement in economic efficiency from the use of environmental tax revenues. See for instance Baumol (1972); Sandmo (1975); Lee and Misiolek (1986); Oates (1993); Bovenberg and de Mooij (1994); Bovenberg and Goulder (1996); Parry, Williams and Goulder (1999).

It becomes apparent that the historical application of MBIs has been dominated by "weak" MBIs which primarily seeked to raise tax revenue. Moreover, "weak" MBIs are often underlain by the CAC approach, which is based on the premise that the sole relationship between ecosystems and the economy is characterised by trade-offs.

Here is therefore a crucial economic concept which policy-makers should understand: this is the ability of economic systems to create conditions where all role players can benefit simultaneously.

In this assignment therefore, we explore EPIs that move beyond the trade-off paradigm. These instruments fall within the category defined by the World Bank (1998) as "strong" MBIs. In the proceeding section, we provide an improved definition for MBIs.

2.1.3 An improved definition of EPIs

Our improved definition of EPIs starts with classifying "weak" MBIs (see above) as fiscal policy instruments and "strong" MBIs as economic policy instruments (EPIs).

Fiscal policy instruments are accordingly defined as environmental policy instruments that seek to raise tax revenue for Government, or impose administration duties and/or fines on resource users.

EPIs are instruments which enable an exchange of good or service to take place in exchange for monetary compensation. This could include a sale of good or service or a trade of commodity.

This proposed definition requires further elaboration as it requires all roll-players to adopt a radical departure from conventional environmental policy instruments.

Central to understanding EPIs (and to achieving true double-dividend outcomes), is the concept of Nash Equilibrium. The Nash Equilibrium is named after its inventor, John Nash, an American mathematician and 1994 Economics Nobel Laureate. In a Nash Equilibrium, role players in a market can each achieve optimal outcomes simultaneously. All role players win, because all get the outcome they desire. In other words, the relationship between the economy and ecosystems is not a zero-sum game. Three sets of role players are relevant here:

- Government acting in the public interest on behalf of public goods, in this case water resources and water – Government desires policy goals to be met, administration costs resulting from oversight functions to be covered and tax revenue to be raised
- Water resource users and water users – These are water users, as defined by the National Water Act (1998)
- Beneficiaries of aquatic ecosystem services – These are all South African who benefit from healthy aquatic ecosystems, whether directly or indirectly.

EPIs would enable all the above role players to benefit simultaneously.

Understanding the relationship between aquatic ecosystems and the economy is also fundamental to understanding EPIs. The MEA ecosystem services framework (Box 1) has provided a radical new way of defining this relationship. The MEA framework enables the analysis of chains of causality linking ecosystems to market role players, and provides fundamental insights into EPIs. This is because it enables the identification and linkage of polluters/impactors, through evidence-based ecological transactions, to beneficiaries of ecosystem services. A highly simplified schematic of these linkages is presented in Figure 2-1 below.

Box 1. Ecosystem services framework

One of most significant weaknesses in the field of environmental economics in the pre-MEA era, was a predominant focus on the direct use values of the environment, with little effort in understanding the indirect linkages between ecological functioning, ecosystem services and the production and consumption of other economic goods and services. In response to these weaknesses, the MEA (2005 and 2007) introduced a radical new framework to the analysis of the interface of the ecology and the economy. So radical is this approach that it has been described by Perrings (2006) as equal in significance (within the field of environmental economics) to the Hartwick rule for reinvestment of Hotelling rents (Crafford et al., 2015).

Central to this framework is the definition of the concept of ecosystem services. MEA (2005) and TEEB (2010) define ecosystem services as the direct and indirect contributions of ecosystems to human well-being. They distinguish between four types of ecosystem services: provisioning, cultural, regulating and habitat services:

- Provisioning services are ecosystem services that describe the material or energy outputs from ecosystems. They include food (from natural resources), raw materials, water and medicinal resources.
- The cultural services include recreation and mental and physical health, eco-tourism, aesthetic appreciation and inspiration for culture, art and design; and spiritual experience and sense of place.
- Regulating services enable ecosystems to continue to produce other direct benefits to humans. Such services may also be interpreted as providing 'insurance' value as they allow the system to continue to function over a range of conditions (e.g. stresses or shocks, often of anthropogenic origin) (Loreau et al*.,* 2002; Simonit and Perrings, 2011). They include regulation of the air quality, carbon sequestration and storage, moderation of extreme events, waste-water treatment, erosion prevention and maintenance of soil fertility, pollination, biological control and maintenance of genetic diversity.
- Habitat provisioning services are defined to provide everything that an individual plant or animal needs to survive: food; water; and shelter. Each ecosystem provides different habitats that can be essential for a species' lifecycle (www.teebweb.org).

(It is to be noted that whereas the first three categories of services are benefits flows from ecosystems and akin to "income statement" items, Habitat provisioning is a natural asset and therefore akin to a "balance sheet" item.)

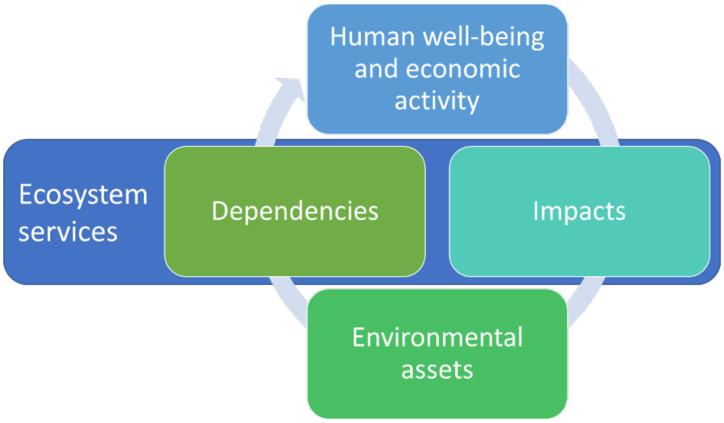

Figure 2-1: The relationships between environmental assets, ecosystem services and human well-being: human well-being, including economic activity, depends on abstractions from and impacts on environmental assets.

Another important concept in defining EPIs, is the concept of transaction clearing mechanisms. In order to ensure that EPIs operate efficiently, they must be effectively regulated (Gensler 2009). EPIs as market-based instruments, thus need to enable regulatory authorities to make adaptive mitigation decisions and consistently improve efficiency. Finding the right regulatory framework and the right amount of regulation requires a delicate balancing act. Under-regulation may allow for fraud and manipulation while over-regulation may be too costly to comply with (Kachi and Frerk, 2013).

Experience from carbon, “acid rain” and ozone markets, provides evidence for market oversight mechanisms. These mechanisms comprise a range of institutional arrangements (policies, protocols, systems, organisations and trading platforms) that we collectively refer to as transaction clearing mechanisms. In finance, “clearing” denotes all activities from the time a commitment in a transaction is made, to the point where the transaction is settled. Mature markets clear transactions through a third party, known as a clearing house (Kachi and Frerk, 2013). Designing for effective EPIs therefore needs to design for effective clearing houses.

Refer to Figure 2-2 for a schematic representation of the relationship between ecosystems, ecosystem services, transactions clearing mechanisms and EPIs.

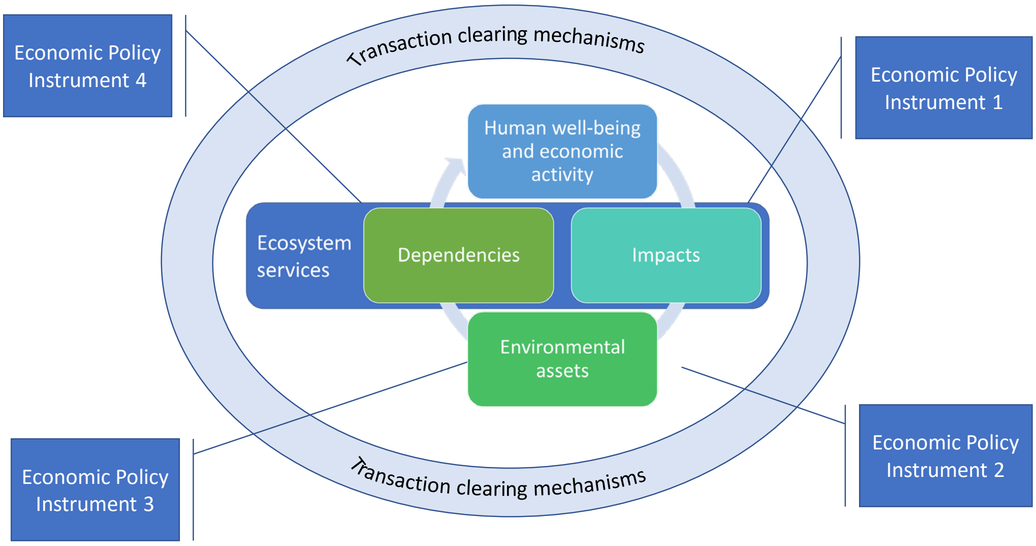

Figure 2-2: The relationship between EPIs, environmental assets, and ecosystem services

Following from the above discussions we therefore propose an adapted policy instrument and EPI typology. It is to be noted that fiscal instruments and EPIs could be designed to work in tandem. It is possible, for instance, that a fiscal policy instrument could serve as a transaction clearing mechanism for an EPI.

This definition modifies and expands Van Nispen's (2011) definition of "Economic" policy instruments. We propose that Van Nispen's definition of "Economic" is more accurately described as "Fiscal", i.e. relating to raising tax revenue. Fiscal instruments can also been viewed as a special category of regulatory instruments, as they are determined administratively. It is to be noted that fiscal instruments and EPIs could be designed to work in tandem. It is possible, for instance, that a fiscal policy instrument could serve as a transaction clearing mechanism for an EPI.

It is further noted that where Government sets use charges, as in the case of raw water, this is not defined as a fiscal policy instrument. The purpose of use charges is to recover costs of capital and operations and they are therefore defined as EPIs.

EPIs therefore are instruments which enable an exchange of goods or services (as envisaged by the MEA) or a proxy for goods or services, for monetary compensation. An effective EPI achieves a win-win situation for all role players. EPIs incentivise investment in aquatic ecosystems. Effective EPIs require as pre-condition, clear and enabling regulatory instruments. EPIs could be used in conjunction with communicative and fiscal policy instruments (see Table 2-2).

Table 2-2: A revised policy instrument typology (adapted from World Bank (1998), Sterner (2003), Van Nispen (2011))

Policy Instrument Type	Incentive-based	Disincentive based	Notes
Regulatory	Prescriptions	Prohibitions	Regulatory instruments remain important for achieving environmental policy objectives. They establish objectives, thresholds and oversight mechanisms. Suitable regulatory instruments are pre-conditions for EPIs
Communicative	Information	Propaganda	These instruments are also referred to as suasion instruments. Their purpose is to share various forms of information with natural resource users / impactors, in order to change their behaviour.
Fiscal	Grants, Subsidies	Taxes	Fiscal policy instruments are environmental policy instruments that seek to raise tax revenue for Government, or impose administration duties and/or fines on resource users.
Economic	Market creating	Not applicable	EPIs enable an exchange of goods or services or a proxy for goods or services, for monetary compensation, including both use charges and market-determined prices. An effective EPI achieves a win-win situation for all role players. EPIs incentivise investment in aquatic ecosystems. Effective EPIs require as pre-condition, clear and enabling regulatory instruments, and could be enhanced by communicative and fiscal policy instruments.

2.2 Benefits of Economic Policy Instruments

The UN Environment (2004) lists a number of benefits associated with EPIs. These benefits were defined by means of evidence-based case studies. The discussions below are case-specific and do not necessarily relate to water.

2.2.1 Reduction in overall cost of achieving emission reductions by providing flexibility

The UN report cites several case studies of air pollution trading systems as evidence of cost reduction. EPIs ensure that the overall economy achieves a desired level of emissions, as opposed to enforcing every polluter to meet a specific emissions level. This allows markets to direct individual firms' emissions. Industries with lower marginal abatement costs tend to control more than required, thus reducing the economy-wide costs of meeting abatement targets. In addition, the ability to purchase emissions credits or to pay pollution taxes as an alternative to installing abatement technology, allows polluters to budget for capital costs associated with compliance requirements. Regulators maintain market oversight. If emissions reductions are insufficient, regulators can adjust the EPIs by increasing fees or decreasing permits.

2.2.2 Encouraged use of innovative abatement technologies

In the case of pollution abatement EPIs, the existence of environmental tax mechanisms endows polluters with a time window in which to select and develop the most appropriate abatement technology. This also means polluters have lower financial risk.

2.2.3 Providing property rights to natural resources to parties who value them most

EPIs that enable fair access (through auction, concession or other means) to public resources enable those who value them most to obtain them. This enables more sustainable use of flow resources (such as water) and more careful use of stock resources (such forests), as well as raising revenues for the government. A risk here is that if access is structured improperly (e.g. rigged auctions or corrupt concessions at a fraction of true value; or when bonding for post-extraction remediation is not done), resource depletion would accelerate.

2.2.4 Self-Enforcement by Aligning Public and Private Interests

EPIs, by their nature, create groups of firms and individuals with vested interests in the proper use of resources. Similarly, groups that are given communal property rights have an incentive to ensure other groups do not encroach on their rights. This enables a more decentralised enforcement system.

2.2.5 Increased Transparency

The costs and rights associated with many EPIs (e.g. marketable permits-pollution tax system) are visible through trading levels, prices and ownership patterns. EPIs do however require effective monitoring, data generation, and enforcement functions.

2.2.6 Cost-Recovery of Public Provision of Services

Administered tariffs or charges, if effectively designed and implemented, can serve to internalise several externalities. In South Africa, publicly owned water resources produce water for different uses. These water resources are in a system that seeks cost recovery. However, in addition to generating cost recovery revenue, the revenue produced can also serve other policy purposes. These can be equity and conservation related.

2.3 Disadvantages of Economic Policy Instruments

Although advocates have expressed a preference for the use of EPIs for environmental policy and consider them more efficient than regulatory instruments (Seroa Da Motta et al., 2004), EPIs can have disadvantages if improperly designed and implemented (Kautto and Melanen, 2004).

A number of disadvantages identified include (BIS, 2011; Sorina-Geanina and Nicoleta-Cornelia, 2011):

- Environmental taxes require complex and costly monitoring
- User charges and fees may encourage illegal discharges
- Tradable permits involve high transaction costs and require the existence of specialised markets and a complex administration system
- It can be difficult to determine the size of the incentive required for changing behaviour
- Change in behaviour may take time, thus there is an uncertainty regarding when the effects will occur
- EPIs provide flexibility, yet after their implementation, it may be difficult for the government to change them
- EPIs may require increasing prices to reduce consumption or reflect scarcity, which may affect equity and hamper their compliance and effectiveness.

Many of the disadvantages listed above stem from poor EPI design. The World Bank (1998) warns that if the primary purpose of EPIs is to raise revenue, it would likely fail. Moreover, the findings of the 2004 UN Environment (discussed in section 2.2) need to be emphasised again, that a "*lack of understanding of how EIs work; political interests that seek to minimize control costs via regulation; and a preference for keeping the status quo*" hampers effective EPI implementation (UNEP, 2004).

Another problem raised by the World Bank, is that public awareness is often low and uncertainty high. These factors, when combined with a mistrust of regulators and/or firms responsible for environmental degradation, would lead to weak or obstructive participation among stakeholders. This poses a real constraint to the rapid implementation of complex EPI mechanisms.

Major industries that can afford to pay polluting penalties can often misuse this instrument as a license to pollute. It is thus important for authorities to make these industries aware of the impact their pollution has on the environment.

In South Africa, major concerns of implementing EPIs are:

- The impact of taxes or charges on the poor and on business (and therefore the possibility of resistance to the instruments from communities and businesses); and
- Uncertainty of whether revenues generated from taxes and charges will be ring-fenced to fund mitigation measures.

2.4 Economic Policy Instruments in the South African Landscape

This section investigates economic policies in South African's water sector that have the potential to mitigate ecological damage. These policies include water tariffs, payments of ecosystem services and conservation credit trading.

2.4.1 Water Tariff

Water pricing is an important economic instrument for enhancing social equity, improving water use efficiency and ecological sustainability, and securing financial sustainability of water utilities and operators. Thus, water pricing can be a powerful management tool to achieve various objectives across the water value chain.

Although water in modern times has become a much scarcer commodity, it remains an inexpensive commodity. However, it is also a social good, and the rights to free basic water and to an ecological water reserve are recognized in South African law. Thus, there arises an apparent conflict between the application of economic charges for water, and the social value of water. This results in disagreement about the “right” way of pricing it.

Different tiers of water tariffs exist across the value chain, spanning raw water charges, pollution abatement charges, bulk water charges, and various water services charges, and they will be further discussed below. The setting of these tariffs is governed and influenced by various policies and processes including the Pricing Strategy for Raw Water Use Charges (DWA, 2007), Parliamentary Portfolio Committee on Water and Sanitation, National Treasury, municipal by-laws and others. The challenge with getting water pricing right is to find a balance between supplying water as a Constitutional right; reflecting the total costs of supply (i.e. sustainable service delivery) and also reflecting the scarcity of the resource itself (i.e. minimise wastage and internalising environmental externalities and opportunity costs).

In 2014, DWS published a water policy position related to economic regulation (DWS, 2014).The position states that "Economic regulation will be applied throughout the water value chain. Scope and functions of economic regulation will encompass the setting of the rules to control, monitor, enforce and/or change tariffs/charge; tariff/charge determination structures and service standards for the water sector whilst recognising and supporting government policy and broader social, environmental and economic imperatives and the function of technical regulation of water infrastructure. To avoid any conflict of interest, real or perceived, water use tariffs will be determined annually by DWS, in consultation with National Treasury."

This policy position is motivated based on the fact that there is currently no coherent economic regulation of the entire water value chain. Accordingly, economic regulation is targeted at specific water management institutions operating in the value chain, including raw water charge setting, water user association charges, bulk water charges, water services charges (water and sanitation) and waste discharge charges and other charges that may arise.

The following Acts provide the legal basis for water tariffs:

- National Water Act (DWAF, 1998) as it relates to raw water tariffs
- National Environmental Management Act (1997) as it relates to dealing with the cost of pollution
- Water Services Act (1997) as it relates to bulk water tariffs and water services
- Municipal Systems Act (2000) as it relates to water services.

The National Water Act (DWAF, 1998) identifies three tiers of tariff for water management (Figure 2-3):

- Tier 1: **Raw water tariffs** – These administered prices are applicable to all water users and comprises the basic input cost of fresh water supply. These administered prices are relevant for services that include the use of raw water from the water resource by bulk distributors, large users and irrigators. This may also include a water quality pricing component for dealing with pollution.
- Tier 2: **Bulk water tariffs** – These administered prices are applicable to all water users who are customers of bulk water service supply entities. These prices include the cost resulting from raw water tariffs. These administered prices are relevant for intermediary water services supplied in bulk (often by water boards).
- Tier 3: **Water services tariffs** – These administered prices are applicable to all water users who are customers of water service supply entities (mostly municipalities). These prices include the cost resulting from raw water tariffs and bulk water tariffs. These administered prices are relevant for water provision and sanitation services to households and other urban and domestic users (usually via a municipality).

Figure 2-3: The three water tariff tiers envisaged by the National Water Act (1998)

2.4.1.1 Tier 1: Strategy for Raw Water Use Charges

The Water Pricing Strategy (DWA, 2007) deals with environmental externalities to some extent through economic charge and water resource management charge.

2.4.1.1.1 Economic Charges

There are two methods of setting the economic charge, and they are: administratively by determining a proxy for the economic value of water, or market-oriented mechanisms.

The administratively determined charge is to be used to promote beneficial use through the reallocation of water to higher value users. This can be accomplished by allowing the transfer of authorisations to use water by trading.

Where amounts of water are still available for allocation after compulsory licenses have been issued and there is competition for using this water, the public tender procedure may be followed.

The purpose of the economic charge is to promote beneficial use through the reallocation of water to higher value users in water stressed catchments. Key benefits of economic charges are that they can allocate water more efficiently and they can assist in curtailing pollution, and this can be achieved without compromising social imperatives. However, two matters arise from the implementation of economic charges: the financial cost and the fiscal impact. The pricing strategy does not differentiate between these but sees accrual of all charges to the National Treasury.

2.4.1.1.2 Water Resource Development Charge and Water Resource Management Charge

Water resource development and use of waterworks refer to the planning, design, development, operation, maintenance, refurbishment and betterment (improvement) of Government Water Schemes (GWS) and schemes to be funded by water management institutions such as the TCTA and WUAs. If water use charges are too low, they will lead to underinvestment, over-consumption and unintended fiscal subsidies. As a result, the water pricing strategy utilises the depreciation, return on assets (RoA), betterments, refurbishment and off-budget funding approach for setting charges to recover capital cost in respect of schemes owned by Government. The funding of off-budget infrastructure developments requires loans. State funding is envisaged to be confined mostly to social, water resource development or betterment projects which conform to the purpose set out in section 2 of the NWA, 1998 and where the demand is not driven by specific commercial water users or sectors.

The Water Resource Management Charge (WRMC) comprises activities that are required to protect, use, conserve, manage and control the water resources and manage water quality located within WMAs, including securing water-related ecosystem services. These tariffs are intended to cover the WRM related costs of CMAs. These costs could include but are not limited to the following activities:

- Planning and implementing catchment management strategies
- Monitoring and assessing water resource availability and use
- Water use allocations
- Water quantity management, including flood and drought management, water distribution control over abstraction, storage and stream flow reduction activities
- Water resource protection, resource quality management and water pollution control
- Water conservation and demand management
- Institutional development and enabling the public to participate in water resources management decision making.

The water resource management charge (WRMC) and water resource development charge, together, should theoretically cover the costs of capital, operations and maintenance. Together they comprise the full cost of water supply. There are, however, additional costs associated with the consumption of water and other aquatic ecosystem services (Crafford, 2012). These include economic externalities and environmental externalities (Rogers *et al*., 1998). Economic externalities refer to the additional cost borne by downstream users and environmental externalities refer to additional costs borne by society, through damage suffered.

2.4.1.1.3 The revision of the pricing strategy for water use charges

DWS released a gazette on revision of the pricing strategy for water use charges (DWS, 2015). The revised raw water pricing strategy provides a combination of national charges and water management area specific charges to facilitate the development of affordable raw water tariffs to all users. The hybrid pricing strategy proposes a number of changes to the current version of the pricing strategy, and these changes aim to increase revenue generated by the Department. The revised pricing strategy also includes the waste discharge charge system (WDCS) which was introduced in the 2007 strategy (DWAF, 2007) (see Figure 2-4). This system is intended to establish an economic instrument for charging waste discharges, or water pollution.

<table>
<tr><td colspan="2">Water Resource Management Charge (capped for certain sectors)</td><td rowspan="4">Targeted subsidies for Resource Poor Farmers</td></tr>
<tr><td colspan="2">Operation & Maintenance (capped)</td></tr>
<tr><td colspan="2">Water Research Commission Charge (resource poor farmers exempt)</td></tr>
<tr><td colspan="2">Depreciation (capped)</td></tr>
<tr><td>RoA (4% of replacement asset value; not charged to irrigation sector</td><td>Capital unit charge (off-budget schemes)</td><td>Water Resource Development Charge (off-budget schemes where loans are paid off</td></tr>
</table>

<table>
<tr><td colspan="2">Waste Discharge Charge (Mitigation)</td><td rowspan="4">Targeted subsidies; e.g. for redress, equitable economic growth, rural development</td></tr>
<tr><td colspan="2">Water resource Management Charge (phased in over 10 years for certain sectors)</td></tr>
<tr><td colspan="2">Operation & Maintenance (capped for agriculture)</td></tr>
<tr><td colspan="2">Depreciation Charge (capped for agriculture)</td></tr>
<tr><td>Future infrastructure build charge (apply to municipal, industrial & mining, high assurance use only</td><td>Capital unit charge (on off-budget schemes)</td><td>Water Research Commission Charge (resource poor farmers exempt)</td></tr>
</table>

Existing Raw Water Pricing Strategy **Proposed Raw Water Pricing Strategy**

Figure 2-4: Changes in charges in raw water pricing strategy

2.4.1.1.3.1 __Waste Discharge Charge System__

This WDCS is based on the polluter pays principle (envisaged in National Environmental Management Act (NEMA)) and aims to internalise the environmental costs of waste dischargers by recovering the costs of mitigating water resource quality impacts of pollution. It further intends to create financial incentives for waste dischargers to reduce waste.

The WDCS is implemented through a catchment level WDCS business plan, developed as part of the water resource management plan (WRMP) in the catchment management strategy (CMS). The business plan articulates technical, financial and management arrangements for the implementation of the WDCS and is developed by the catchment management agency (CMA), with support and oversight from DWS. DWS will develop a consolidated WDCS plan, based on the catchment business plans, and will negotiate the consolidated plan with National Treasury.

According to proposed strategy WDCS is intended to:

- Promote the sustainable development and efficient use of water resources
- Promote the internalisation of environmental costs by waste dischargers

- Create financial incentives for waste dischargers to reduce waste and use water resources in a more optimal manner

- Recover costs associated with mitigating resource quality impacts of waste discharge.

2.4.1.1.3.1.1 Application of WDCS

WDCS is based on a linear relationship between load and charge (i.e. a flat charge rate is applied). In other words, the charge increases by constant increments with an increase in discharge load.

The WDCS comprises two categories namely the **waste discharge levy**, that will provide a disincentive for the use of the water resource as a means of disposing waste and the **waste mitigation charge**, which will cover quantifiable costs of mitigating waste discharge related impacts.

The waste discharge levy is an unreturned payment in that it does not recover any direct costs, nor is it related to a particular service received and as a result, this charge will generate surplus revenue which could be used for a number of uses, through a process of implicit assigning and budgetary allocation. The levy is based on monitored discharge load, given that the charge seeks to change actual discharge load. This also means that where dischargers are reducing waste load at source, that reduction is reflected in a reduced charge.

Waste mitigation charge is intended to cover the costs of mitigation measures undertaken in the water resource and will be applied in cases where it is more economically efficient to reduce load within the resource than reducing discharge load at source. As such, the Mitigation Charge is a user charge to recover the costs of mitigation measures deployed in the resource. There are four categories of Mitigation Charge:

- Mitigation through removal of load from the resource, including a regional mitigation scheme or infrastructure or a regional mitigation project

- Water resource system operation for the dilution, blending or purging of poor quality water

- Mitigation for treatment costs downstream

- Treatment at source, in order to apply the most cost-effective treatment options to a limited number of dischargers in a catchment Institutional arrangement.

2.4.1.2 Tier 2: Bulk Water Pricing Structure: Overview

The Water Boards are regulated by the Minister in terms of the Water Services Act, 1997 and the Public Finance Management Act. As a result, DWS has published a guideline for Water Board tariffs. According to this guideline, bulk potable water tariffs must achieve and maintain the pre-negotiated financial targets and capital structure of the water board as expressed in the Shareholders Compact, after providing for expenses, capital investments and loan repayments and for a reasonable amount of flexibility through providing for contingencies and, where applicable, for agreed dividends to the National Revenue Fund.

These tariff guidelines must also be read in conjunction with the National Treasury "Distribution Policy for Government Business Entities (NT, 2016a)" and with the National Treasury "Capital Structure Policy for Government Business Enterprises (NT, 2016b).

2.4.1.3 Tier 3: Municipal Water Services Pricing structure: Overview

The Constitution of South Africa provides the national government, subject to section 44, with legislative and executive authority to oversee the effective performance by municipalities of their functions, including water and sanitation limited to potable water supply systems and domestic wastewater and sewage disposal systems.

In October 2015, government published a gazette on norms and standards in respect of tariffs for water service authorities and bulk water service providers (DWS, 2015). This revised document pays attention to water challenges in South Africa and the charges are drought and seasonal charges. This document also recognises that the state of water treatment works needs improvement by having a charge that will be able to recover operations and maintenance costs.

2.4.1.3.1 Drought tariff and seasonal charge

The new policy recommends that a Water Service Authority may implement drought tariffs or activate other conservation measures such as imposing water restriction including:

- If drought tariffs have not been approved as part of the water supply tariffs, a Water Service Authority must follow processes described in the norms and standards before implementing drought tariffs
- Authority must use a tariff that minimizes the economic and social costs related to water conservation measures
- Authority must use a tariff that will encourage consumers to reduce consumption to the sustainable drought level with little time lag, by for example charging a penalty volumetric tariff or additional fixed tariff for consumption above the desired sustainable consumption.

2.4.1.3.2 Revenue required to provide water supply on a sustainable basis

A water service authority must set its water supply tariffs so that its water supply revenue inclusive of all other transfers and grants allocated to water supply services is sufficient to recover:

- All reasonable costs directly and indirectly associated with the operation, maintenance, refurbishment and development of water services, water services customer care and all costs associated therewith

- Payment required to redeem its water services related loans over a reasonable period

- A net surplus of a minimum of 6% per annum on revenue.

A water service authority must at all times budget separately, prepare separate financial statements, and set tariffs separately for its water supply services function.

2.4.1.3.3 Industrial Wastewater Charge

Industrial effluent has generally been regarded as an unimportant and troublesome by-product of a manufacturing process. In a few cases, industrial effluents contain useful resources that can be viably recovered. In South Africa, industrial effluent discharge is ultimately regulated by DWS. If an industry is located within a municipal boundary, the industrial effluent will most likely find its way to the municipal sewage treatment plant. Traditionally municipalities have imposed minimum discharge standards. These standards were primarily imposed to protect the sewage network. For example, low pH was discouraged to prevent corrosion of cement pipes. A limit was imposed on suspended solids to prevent clogging of pipes. Flammable liquids were not allowed to be discharged for obvious reasons. Other standards, such as minimum COD limits were imposed to avoid over-burden of the local treatment works.

Due to public and environmental pressure, DWS has become stricter with municipalities that do not meet minimum discharge criteria. As a consequence of this pressure, municipalities have had to examine the capacities of their treatment plants, sewage networks and customers.

The structuring of wastewater charges is a fairly complex process, dependent on several contextual considerations including (DWAF, 2007), the type of water users occurring within a given municipality (and the resulting pollutants that enter the system), the classification status of the water resources intended to absorb any untreated effluent and their associated ability to absorb it; also seasonal variation thereof, and the overarching objective of protecting water resources necessitates charge considerations that promote the prevention of water resource degradation rather than remediation of polluted/damaged water resources.

The development of an effective wastewater treatment charge is thus dependent on an intimate understanding of the various factors that determine how it should be structured. A lack of understanding in any of the underlying contextual considerations (e.g. monitoring; resource quality, resource quality objectives) results in a sub optimal wastewater treatment charge.

A significant amount of work has been done to delineate the complexities of setting wastewater treatment charges, most notably and most recently by Hosking et al. (2011). However, despite the availability of such resources recent investigations as part of the DWS Olifants Water Resource Classification System (DWA, 2013) have revealed that these proposed charge structures are not being applied at the municipal level, especially in municipalities that struggle to achieve Green Drop certification.

This suggests that the reasons for the lack of implementation of new and relevant charge structures within municipalities is most probably due to a lack of understanding in one or more of the contextual considerations that are required to set the charge.

2.4.2 Payment for Ecosystem Services

The FAO (2007) has defined Payment for Ecosystem Services as follows: "PES transactions refer to voluntary transactions where a service provider is paid by, or on behalf of, service beneficiaries for agricultural land, forest, coastal or marine management practices that are expected to result in continued or improved service provision beyond what would have been provided without the payment."

Internationally, various payments for ecosystem services (PES) schemes have been attempted as a method for internalizing ecosystem benefits into the economy. There have been many case studies and projects internationally, and some in South Africa, and the WRC has also done investigations into the role of PES in Catchment Management Agencies (CMAs) (Pearce, 2014). PES are payments offered by beneficiaries of ecosystem services to service providers who provide some form of management of the ecosystems that produce those ecosystem services. The current form of PES has had very limited success, mostly because it requires the service provider, usually a community, individual or organisation to either cede their rights to land or a natural resource, or to refrain from a potentially damaging or polluting land use or a natural resource use; and in return receives a payment. These voluntary transactions have been problematic because it puts livelihoods at risk, ecosystem services are often not well-defined nor measurable, and tenure is of a public nature, and therefore not secure nor exclusive.

However, UNEP (through the UN Programme on Reducing Emissions from Deforestation and Forest Degradation (UN-REDD programme) and Kenya Forest Service) and the USAID Planning for Resilience in East Africa through Policy, Adaptation, Research and Economic Development (PREPARED) programme, have in recent years made some breakthroughs in PES case study projects in East Africa (in Kenya and around the Lake Victoria Basin) by recognizing two principles, that:

- Linking ecosystems to economic systems requires a common and measurable currency, and
- Adequate scientific evidence between the common currency and associated ecosystem services (multiplier effects) is required.

Analysis of the outputs of these UNEP and USAID initiatives reveals two examples of common currencies: land and carbon. Both are measurable, both have demonstrated scientific linkages with ecosystem services and where the link between the service provider and the beneficiary is direct and secure, PES transactions have been possible. The problem for the South African context however, is that these successes were possible in settings unique to East Africa, i.e. in mega protected areas with extremely high tourism potential (e.g. Masai Mara) (UNEP,2012).

The concept of a common currency is useful because UNEP has demonstrated the existence of ecosystem services multiplier effects related to such common currencies. However, PES systems of the conventional nature described above are not suitable, and neither is a unique currency enough.

In order to ensure water security, decision makers can not only penalise harmful behaviour, they can also reward beneficial actions. The most common way to do this is through subsidies. Payments for Environmental Services (PES) are a specific type of subsidy.

In the context of water, PES can be used to persuade users of land or other natural resources to modify their behaviour so as to protect and enhance water resources (e.g. shifting to organic farming, converting arable land to pasture, planting trees). PES may compensate them for the extra effort and/or financial cost involved in changing their behaviour.

The funding for PES may be provided by governments, international agencies, local communities, water companies, hydro-power producers, flood protection agencies, or private companies, depending on the type of benefits expected and their impact. In some cases, the cost can be passed on to final consumers (e.g. in the higher price of products cultivated organically or sustainably).

In many circumstances, PES may be a more effective and efficient method of managing water resources compared to its alternatives. These typically entail major outlays on water treatment, flood control, and the development of new sources. For example, improving the quality of raw water is normally preferable to investing in costly treatment works, and may be vital for the preservation of brand image (Vittel case below).

There are at least three prerequisites for implementing PES: an effective supply and demand of the targeted ecosystem services; supportive intermediary organisations to facilitate the PES mechanism; and supportive national conditions (e.g. policies that promote secure property rights and market exchange).

PES recognises the fact that the task of environmental stewardship is widely dispersed throughout society. Getting it done at the source by enlisting key stakeholders can be effective, efficient and fair. Even though PES was not intended to reduce poverty, there are a number of ways that PES programmes can be set up to benefit the poor. For example, direct benefits may be generated through payments to poor people who are the suppliers of ecosystem services. These payments can be monetary or non-monetary. PES programmes may additionally bring indirect benefits such as social capital and political voice, or technical training.

Some examples of PES in action:

The city of New York opted to support farmers carrying out watershed protection upstream in the Catskill Mountains in order to reduce the high cost of treating water downstream closer to the city (Appleton, 2002).

Nestle, a multinational drinks company, operates a scheme for subsidising farmers to avoid the use of nitrates in the area from which its bottled water Vittel is drawn (Perrot-Maitre, 2006).

In Quito, Ecuador, and in several smaller cities in Honduras and Costa Rica, the water utility and electric power companies pay local people to conserve the watersheds from which water is drawn (Pagiola et al., 2008).

In Venezuela, the power producer CVG-Edelca pays a proportion of its revenues towards the preservation of the Rio Caroni watershed (Dasos Habitat Foundation, 2017).

Experience with PES shows that it is most likely to succeed where the following conditions are present (Fripp, 2014):

- There is a clear demand (need) for ecosystem services, which have financial value to one or more stakeholders
- Provision of ecosystem services is threatened
- Specific resource management actions offer feasible solutions
- Effective brokers or intermediaries exist
- Resource tenure is clear, and contracts can be enforced
- Outcomes of actions can be independently monitored and evaluated.

2.4.3 Conservation Credit Trading

Given the degraded state of much of the country's wetland resources, the increased threat to wetland resources in certain regions and the growing application of questionable approaches to offsets measures for wetlands, there is an urgent need to develop a policy that will mitigate these impacts, and this can be done through conservation credit trading. Although conservation credit trading in the South African context is fairly new, there have been some pilot projects that show successful wetland mitigation banking implementation.

Currently, wetland rehabilitation through offsets is undertaken on an individual basis in response to permitting conditions. Wetland offsets are designed for measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken (BBOP, 2009).

Conservation credit trading is the preferred mechanism for establishing compliance with regulatory requirements (and national policy). It involves the establishment of new wetland areas, or "banks," in advance of anticipated losses (SANBI, 2008).

The usefulness of a bank to decision makers and developers is therefore greater where the demand to develop wetlands is high, i.e. where there is a sector that impacts wetlands and there is projected growth in that sector in a catchment. However, the majority of wetland habitats in South Africa are located in predominantly agricultural areas yet, the presence of wetlands on agricultural land do not contribute to the economic value of land. Therefore, for wetland banking to be viewed as an economic policy instrument, more valuation studies on wetlands are necessary.

Conservation credit trading has a number of advantages over individual fragmented wetland offsets because of the ability of mitigation banking programs to (SANBI, 2008):

- Reduce uncertainty over whether the compensatory mitigation will be successful in offsetting project impacts
- Assemble and apply extensive financial resources, planning, and scientific expertise not always available to many permittee-responsible compensatory mitigation proposals
- Reduce permit processing times and provide more cost-effective compensatory mitigation opportunities
- Enable the efficient use of limited agency resources in the review and compliance monitoring of compensatory mitigation projects because of consolidation.

Additionally, banking represents an increasingly important economic component of the environmental consulting sector, showcasing the synergies that can arise between effective environmental protection and economic expansion.

2.5 EPIs Implemented Internationally

2.5.1 Australia

Australia has been using water trading as a mechanism for reducing the economic opportunity costs of water. The policy imperative was to move water use away from rice production to higher value output water uses.

In the 1980's there was a fundamental shift in focus with regards to water management. Government was less willing to fund large scale water infrastructure projects, the agricultural sector was becoming increasingly exposed to international competition and there was an increasing awareness of the impact of large-scale water storage and use on the environment (NWC, 2012).

Water resources in certain catchments were becoming fully or even over allocated leaving agriculture and other economic sectors vulnerable to shocks in the available water supply. This resulted in a shift away from the augmentation of existing water supply schemes towards making better use of the existing water entitlements. This led to the introduction of a water trading system in 1983 whereby water entitlements could be traded between water users.

One of the primary advantages of the system of tradable water entitlements was that it allowed for the overall extractions of water to be capped, meaning that the only way to obtain additional water rights was to purchase them from someone else.

The underlying motive was to increase efficiency as well as equity in water allocation whilst simultaneously preserving the environment for coming generations. Fundamentally, this enables water allocations to move to relatively higher valued uses which reduces the opportunity costs associated with the use of water and results in a benefit for the overall economy (NWC, 2011).

The price signal that resulted from the trade of water entitlements encouraged irrigators to use water and other inputs in a productively efficient manner. Water prices signals during the droughts of the early 90's encouraged irrigators to determine exactly how much water their crops required during particular phases of their growth cycles and to invest in technologies that improved the efficiency of their water use.

The result of these new approaches saw shifts on a regional scale with large volumes of water moving away from flexible production systems such as the rice growing areas of New South Wales towards long lived perennial horticultural investments in South Australia (NWC 2012).

The trade of water entitlements also saw water move from the agricultural sector to urban centres as well as the environment.

The system has encouraged the development of high value irrigation enterprises whilst allowing for adjustments to production to be made more easily. This allows market exit and retirement decisions to be made by producers as the sale of water entitlements allows individuals to finance their exit from the industry, or simply to negotiate a period of change.

This case provides evidence that water trading can reduce economic opportunity costs. The benefits thereof accrue indirectly through increased Gross Domestic Product (GDP) and to the fiscus through higher tax revenues associated with increased production. The costs of implementing water trading schemes would most likely accrue to CMAs and need to be recovered either through the raw water tariff, or through a mechanism ring-fenced within the water trading scheme. In addition, it may also be possible to capture the resource rent associated with these transactions – these rents would likely accrue to National Treasury.

The NWA makes provision for transfer of water use authorisations (Section 25). However, since 1998, many instances of water trading have emerged that have undesirable unintended consequences, which were contrary to water equity principles. This resulted from regulatory powers in water allocations to water management institutions and limited oversight mechanisms for DWS. Two key undesirable, unintended consequences that arose from this were: (a) that the allocation of water may take place without Ministerial consent; and (b) that water, because it is a scarce good, becomes commoditised, and water traders use it as a means to make profit, at the cost of users.

2.5.2 Israel

During the past five decades, Israel has developed broad and diverse policies to protect and preserve the environment, based primarily on CAC regulations, and began to apply EPIs in environmental policy only in the last 2 decades. Yet this approach is quickly gaining momentum, as use of EPIs is increasingly viewed as a legitimate option in environmental policy (Bank of Israel, 2009; MoEP, 2008). In the past, due to the absence of specific laws and regulations, restrictions on pollution emissions were implemented through conditions stated in the business license of the relevant factories. This situation was characterised by a fundamental lack of consistency and transparency, with the absence of a planned, guided policy. In addition, company owners could use their economic and political influence to obtain less stringent conditions in their business license.

According to Palevitch et al. (1998), one major concern that delayed the use of EPIs in Israel's environmental policy is related to concerns regarding the implications on the economy's competitiveness in the international market, and fear of affecting the competition ability. In addition, policy makers were influenced by the concern that the implementation of such measures would lead to a decline in GDP growth. Additional possible obstacles are policy makers that have an in-depth knowledge and practice of using regulations; a lack of economic expertise within the national administrations; and the existence of interest groups that support/ oppose certain policies and try to influence them.

Over the years, many studies have been carried out on the economic implications of environmental protection in Israel, including a study analysing the economic aspects of environmental liabilities (Barkat et al., 1996). As a result, awareness to the problems associated with the existing situation increased and Israel went through a conceptual revolution regarding the use of EPIs in environmental policy. A particularly significant change occurred in 1995, when the minister of environmental protection established a public-professional committee (Mingelgrin Committee) in accordance with the Government Decision 6043, to set regulatory procedures to protect the environment. In 1997, the committee published its recommendations concerning the importance of establishing a structured and transparent process for setting environmental standards in Israel (MoEP, 1997). According to the committee's recommendations, one of the main principles is that the process should be based upon an economic methodology, by conducting a cost-benefit analysis (CBA) in order to determine the optimal standard from the perspective of the national economy. Since the committee published its recommendations, extensive regulation has been established. In particular, CBA were carried out to determine the optimal policy on many environmental issues. Indeed, this procedure has become widely used for setting standards and regulations in many areas, such as water, energy and transportation.

2.5.3 Lessons Learned

The prospect for EPIs needs to be understood in the context of the country’s water development.

In Israel, a combination of increasing block tariffs, excess water use fines, production levies, tariffs to finance artificial recharge, subsidies for well rehabilitation, long term commitments for purchasing water produced by new sources and a close to full cost recovery water price system has proved to be a successful policy reform for addressing severe water scarcity and providing adaptive mechanisms for coping with drought. Water prices and their role can only be understood in the context of an IWRM system and in the context of the different role that water has played in economic development. In Israel, this has shifted from an initial stage where the dominant objective was food and water security to the present when environmental sustainability is a paramount concern.

Consistent policy actions can result in the successful implementation of different EPIs making for example water trading the best alternative in some places (as in the Murray-Darling basin in Australia) and command and control and prices in others (as in the case of Israel).

The framework conditions and the political process are more important than the instrument itself in explaining the success or failure of each case. This might also explain why EPIs may deliver benefits for the green economy even when their implementation fails. For example, in the case of the over-consumption tax in Israel (which ultimately failed), the initiative helped raise public awareness and discussion of water challenges which can assist in the search for better social responses.

Various international examples of incorporating the economics of aquatic ecosystems and biodiversity into water pricing do exist, these are however not easily transferrable to other countries. There are three key reasons for this: firstly, countries and their river basins have unique socio-ecological settings comprising unique ecosystems, ecosystem services bundles and socio-economic conditions; secondly, economic realities and water policy imperatives create unique supply and demand settings; and thirdly, countries are characterised by unique institutional settings.

3 UNDERSTANDING LIMITATIONS

Prior to proposing a solution of internalising EEB into water resource management, it is important to understand the scope of the problem. The following section describes the process followed to identify key limitations to effective use of current policies that tends to mitigate ecological damage in South Africa.

The literature review in the above section formed a key source of evidence and it provided a general framework from where the line of questioning would be guided towards challenges and gaps appropriate to the study.

It was noted through literature review that the South African water sector has enabling legislation for mitigating ecological damage.

3.1 Approach and Methodology

The process followed a participatory, qualitative research approach and gathered evidence from literature and key stakeholder interviews.

A mapping exercise of identifying major impactors was done in order to identify the regulators in water management relating to the use of policy instruments in South Africa (see Table 3-1 and Table 3-2). A sample of purposely selected key informants were interviewed through face-to-face and telephonic interviews to provide a deeper understanding of the major challenges in the water related economic policy landscape in South Africa. Although, the focus however, of the assignment was predominantly on EPI's, the entire policy instrument landscape was assessed with respect to risks to aquatic systems and aquatic ecosystem services.

Table 3-1: Impactors and Regulators of Water Resource Management

Impact	Impactor	Regulator / Implementer
Water use	Raw water, bulk water and water services users	DWS, WSAs / DWS, CMAs, WUAs, Water Boards, WSAs
Habitat provisioning and ecological infrastructure	Cumulative degradation	DWS, WSAs / DWS, WSAs
	Mining; construction; urban developers; agriculture	DWS, DEA, DMR, WSAs / CMA, WSAs, Industry
Water purification and waste assimilation	WWTWs; Industries (e.g. mining, manufacturing)	DWS/CMA
		WSAs/WSAs
		DWS/Industry
	Agriculture runoff (polluted)	DWS / CMA, Agriculture

Table 3-2: Organisation and personnel interviewed

Organization	Target Interviewee
Department of Environmental Affairs (DEA)	Chief Directorate: Integrated Environmental Management- Ms Dee Fischer
Department of Water and Sanitation (DWS);	Chief Director: Water Trade Entity (WTE)- Norman Mudau
Department of Water and Sanitation (DWS);	Acting Chief Director: Economic and Social Regulation- Ms Sizani Moshidi
Catchment Management Agency (CMA);	CEO: Breede-Gouritz CMA- Phakamani Buthelezi
Private Sector	Mr Mark Botha (Independent Environmental Services). Has worked in making financial provisions for environmental mitigation.

In total, 5 interviews were conducted. Interviews were formally planned and took place by appointment. The stakeholder consultation took the form of semi-structured, open-ended expert interviews. Each interview started with familiarising the interviewee with the study, its purpose and key terminology. Thereafter the interviewee was given the opportunity to share their own assessment of the challenges and possible solutions related to water resource management. The interviewer, where necessary, asked for elaboration or explanation with follow-up questions. The interviewer further used research questions to pursue additional lines of enquiry that may not have been raised by the interviewee(s). Interview feedback was captured for analysis. On average, each interview had a duration of 1 hour.

3.2 Gap Analysis Results

The information gathered from interviewees together with literature was analysed and major issues that emerged included:

1. Financial constraints
2. Cumulative water pollution
3. Water pricing strategies
4. Complying with legislative requirements
5. Consumer behaviour
6. Roles and responsibility.

These issues are discussed in brief below.

3.2.1 Financial Constraints

3.2.1.1 Water Trading revenue collection challenges

The Water Trading Entity (WTE) reports to the Accounting Officer of DWS. The entity manages the recovery of usage costs to ensure the long-term sustainability of the country's water resources. The entity has two components: water resource management and infrastructure management. The water resource management component oversees the management of water quality, conservation and the allocation of water through Catchment Management Agencies where these are established and operational. The infrastructure component oversees the operation and maintenance of existing water infrastructure as well as the development of new infrastructure.

WTE has major constraints in revenue collection which include:

- There are no proper billing systems in place especially for municipalities in the rural areas where they have communal taps.
- Revenue collection is inefficient because there is no integration with the deeds office. When a farmer sells the land, they don't have to report to DWS and the department ends up billing the wrong person and that person usually will not report as they know they cannot be prosecuted. Water use License application not being electronic exacerbates the problem.
- The department currently has historic debt which means that they have negative equity. This shortfall results in the department not being able to have sufficient funds to rehabilitate degraded water resources. According to the interviewee, the waste discharge charge to be implemented by the department will not be able to account for communities on riparian zones who discharge waste directly to the river and leakages from sewage pipes.
- Non-revenue water due to aging infrastructure.
- Enforcement and compliance not properly done.

Lack of communication between WTE and Chief Directorate: Economic and Social Regulation makes it difficult for some of the WTE challenges to be resolved. WTE is currently unable to recover their costs, resulting in the department using revenue collected as running costs and not allocating appropriate funds for rehabilitation of ecological infrastructure.

3.2.1.2 Operation and Maintenance (O&M) Costs of WSA water infrastructure

The Water Supply and Sanitation Policy White Paper (DWA,1994) has a significant focus on the economics and financial aspects of water, envisaging that water services should be self-financing at a local and regional level. It mentions at least three principles to this end, namely:

1. That water has economic value, and that service provision has to reflect the growing scarcity of good quality water in South Africa in a manner which reflects its value yet does not undermine long term sustainable and economic growth (South Africa, 1994)

2. That there has to be equitable regional allocation of development resources, as limited resources are available to support the provision, and thus basic services should be equitably distributed among regions, taking account of population and level of development (South Africa, 1994)

3. That the cost of use must ultimately be recovered from the user (this does not exclude the use of pro-poor subsidies to, for instance, indigent households), and is a central principle to ensure sustainable and equitable development, as well as efficient and effective management.

The NWRS (DWS, 2013) reinforces this earlier policy position though under the heading "*Financing the water sector*". It envisages "*optimal investment*" to ensure a financially sustainable water sector. This would include attracting funding for water sector projects from both public and private sectors, reflecting the economic value of water, both in the use of water and the prevention of water pollution, yet ensuring that costs recovered through regulated water prices are pro-poor while promoting water conservation and deterring pollution. The NWPR further envisages the establishment of an Economic Regulator for water.

The Strategic Framework for Water Services (SFWS) (DWAF, 2003) introduced a more detailed financial framework for the water services sector of South Africa. This framework outlines the various funding sources for provision of water services in the country. Two funding mechanisms are the chief source of funds to WSAs for provision of water services, namely (1) payment for the services by the consumer and (2) subsidies from national government to eradicate infrastructure backlogs. National government subsidies mentioned by the SFWS are the Municipal Infrastructure Grant (MIG) and the Capacity Building Grant. MIG is provided to WSAs to address the universal service obligation (a policy priority in South Africa) of providing all South Africans with at least a basic water facility and a basic sanitation facility. In providing these services, the WSA could also utilise consumer payment and other financial resources to cross-subsidise water services within their jurisdiction.

The SFWS also introduced a free basic water services policy. The Free Basic Water Policy (FBWP) has the purpose, according to the SFWS, *to assist in promoting sustainable access to a basic water supply by subsidising the on-going operating and maintenance costs of a basic water supply service.* The FBWP strengthens the 'user pays' principle, as the free component of water is only to provide a volume of water to the poor for a water supply necessary to sustain life. A free basic water supply is defined as 6 kl per household per month. The SFWS indicated that the free basic water services could be financed from the local government equitable share as well as through cross-subsidisation between users within a system of supply or within a water services authority area where appropriate (DWAF, 2003). This basic level of water is provided free to all households that earn less than the poverty line, which is defined as a monthly household income of R2 300 per month by National Treasury in 2013 (South Africa, 2013).

WWTW infrastructure and operations in South Africa faces significant challenges. DWS's Green Drop assessments have served to highlight and communicate these challenges, shifting focus to the sustainable operation, maintenance and rehabilitation of existing water and wastewater schemes. In spite of the current efforts to implement a cost-reflective payment system as well as the existing grant funding, municipalities do not have proper O&M budget to address upgrading and O&M of WWTW infrastructure.

This also forces many municipalities to cross subsidise services to manage cash flow or to fund other non-revenue services.

3.2.2 Cumulative Water Pollution

The 1997 White Paper on a Water Policy for South Africa has as one of its key principles (principle 16) that water quality management options shall include the use of economic incentives and penalties to reduce pollution; and the possibility of irretrievable environmental degradation as a result of pollution shall be prevented (South Africa, 1997).

Various interviewees in this scoping study articulated a number of pollution issues, including contamination of various key rivers and dams, dysfunctional waste water treatment systems and poor maintenance of the infrastructure; spillages from municipal sewerage reticulation systems and discharges from mines and industries. Point source pollution is highlighted as a major cause of concern in effective water management. Interviewees indicated that there are cases where mine and agricultural waste are disposed directly into main sewer lines.

Another water pollution aspect of concern is the cumulative effects of pollutants. Nutrient pollution in particular is predominantly caused by high volumes of WWTW effluent (point sources) and sewage spills (diffuse sources).

The NWRS specifically confirms environmental degradation and resource pollution as one of the key challenges facing the country, resulting from (DWA, 2013):

- Pollution from overloaded WWTWs
- Discharges of urban and industrial effluents to the environment
- Irrigation return flows with high salinity concentrations
- Wash off and leachate from mining operations
- Wash off from areas of human settlement with inadequate sanitation.

The Hartbeespoort Dam, for instance, is heavily impacted by various effluent pollutants. Here, eutrophication is a significant hazard. Eutrophication is the enrichment of fresh water bodies by inorganic nutrients. It may occur naturally, but it is mostly the result of human activity. The main nutrients responsible for eutrophication are phosphates and nitrates, with the phosphates being the main cause. The major sources of nutrients are anthropogenic and result from point sources in the form of

WWTW effluent and diffuse sources such as sewage spills and fertiliser runoff. Eutrophication is particularly evident in slow-moving rivers and shallow lakes, such as the Hartbeespoort Dam, where it will cause various problems if not controlled.

A decrease in biodiversity may result as eutrophication kills the more sensitive species of fauna and flora. One effect of eutrophication is that it causes the loss of desirable fish species. In addition, direct spills of sewage deplete the oxygen in the water killing the organisms living there.

Excessive growth of algae or plants is a problem. The most common algae in eutrophic systems are the blue-green algae. These cause bad odour problems and have the capacity to develop toxins. There have been a number of cases of livestock and wildlife deaths through ingesting algal toxins. Microsystin, one of the toxins generated by the blue-green alga Microcystis, is a small, stable molecule that does not break down easily. This renders the water unsafe to drink, and people especially at risk are those who rely on the water directly from the environment for their needs. Excessive plant growth, water hyacinth being the most common in Hartbeespoort Dam, covers the water surface with a dense growth, preventing light from penetrating the water. As it dies the dead material sinks to the bottom and microbial action uses the available oxygen, killing off the other organisms which contribute to the stability of ecosystems.

Another problem relates to treating of water for potable or other uses. Algae causes blockages in WTW systems, and in addition, in order to render water from a source contaminated with blue-green algae safe it is necessary to use activated carbon. This increase water treatment costs.

Eutrophication negatively affects the aesthetics of a water body thereby negatively influencing property value. Not only does the water look unattractive but blue-green algal blooms may cause the water body to develop a bad odour. Dense growths of plants such as hyacinth on the surface affect the aesthetics. These effects will not only impair the recreation experience of users, but where residential or business facilities are adjacent to the water the value and the experience of using the facility would be impacted.

While high nutrient loads on its own is not a concern to irrigated agriculture, the possibility of residual pathogens or cyanobacterial toxins on the irrigated produce are a concern that irrigators address before marketing their crop (Mitchell and Crafford, 2015).

The above risks may result in economic costs. Graham et al. (2012) showed that severe eutrophication not only increased costs of water treatment, but also increased the cost to agriculture (livestock health and crop losses), caused a decline in value of property adjacent to the water body and lead to a substantial decline in recreation services.

In general, eutrophication needs to be prevented at source and its effects mitigated as it has a negative impact on the health of the people and the national economy.

3.2.2.1 Water Pollution from Acid Mine Drainage (AMD)

South Africa is well endowed with vast mineral resources and the wealth created through mining, particularly gold mining, funded the initial development of the country. However, closure of mining operations on the Witwatersrand since the 1970s and the subsequent termination of the extraction of rising underground water from mines have become an important national concern (Council for Geoscience, 2010). AMD causes toxic environmental pollution, corrosion and acidic pH from the sulphuric acid formed. Rising water levels in the mine voids lead to serious safety risks to deep underground mining operations and pose risks to safety and property on the surface in the surrounding areas of the mines (Durrheim et al., 2006; Goldbach, 2009).

Pumping and treatment process introduced through the immediate and short-term solution only neutralise AMD’s high acidity and remove metals (notably iron) carried in the water; this partially treated AMD is discharged into natural watercourses still heavily laden with salts results in raising saline levels in the riverine system.

A range of statutes has been enacted to give effect to the constitutional objectives, including the NEMA, which regulates the protection of all environmental resources, including water; the WSA, which regulates access to potable water supply services; the NWA, which ensures the management, protection and conservation of water resources; and the National Environmental Management: Waste Act (NEM: WA), which provides for the management of waste. South African law also provides for the regulation of mining in the Mineral and Petroleum Resources Development Act (MPRDA). Collectively the statutory framework includes detailed pollution prevention, minimisation and remediation provisions as well as liability provisions. Liability in the context of AMD is demarcated by means of detailed statutory provisions ranging from framework to sectoral legislation which relates to mining and water resources, and which would be applicable to the issue of water and AMD.

Importantly, DWS issued section 19 notices to mines to take steps to prevent pollution at their own cost. Ensuring liability for environmental damage and pollution forms part of the state's duty to manage the assets of the trust, as it were, as far as its duties as South Africa's public trustee over water resources are concerned. It seems as if South African courts are inclined to support environmental governance departments in this respect.

The problem is that the liability framework, even though it operates retrospectively and despite its comprehensiveness, cannot hold the private sector to account for any damages caused by AMD if the owners of the mines do not exist anymore and AMD in the western basin is largely caused by defunct mines which are ownerless.

What the law can do is to provide a comprehensive liability regime today and applicable to the future so that the current AMD disaster is not repeated many years from now. The financial remedies must be sufficient to address concerns of (future) temporal scale and at present they are not, both in terms of their quantum and the future time frame they apply to. Although this financial provision has to be revised on an annual basis, the calculation of the quantum of financial provision is determined by a departmental guideline that makes provision for three years of post-closure maintenance, which is wholly insufficient considering the pervasive pollution and damage-causing characteristics of AMD.

3.2.2.2 Water Pollution from Wastewater Treatment Works (WWTWs)

Government introduced the Green Drop certification programme, a suasion-type incentive programme for WSAs, to encourage and monitor effectiveness of wastewater treatment in WSAs. In 2010, the Department of Water and Sanitation (DWS) published the first national Green Drop Report, based on assessments done in 2009. Only 7.4% of the assessed plants achieved Green Drop certification. A subsequent Green Drop assessment was done in 2011 and only about 10% of WSAs assessed achieved a Green Drop score exceeding 80%. No other Green Drop reports have been released since then. Effective wastewater management is a requirement to safeguard South Africa's water resources and their associated economic, social and environmental benefits. The Green Drop assessments demonstrate that most wastewater treatment systems fail to effectively treat wastewater and therefore serve as a significant source of pollution. There are numerous reasons for this, one of which may be partly due to ineffective wastewater charges.

3.2.3 Water Pricing Strategies

3.2.3.1 Raw Water Pricing Strategy

A potential conflict between the application of economic charges for water, and the social value of water, has to be managed. This conflict was described to some extent by Adam Smith in An Inquiry into the Nature and Causes of the Wealth of Nations: A Selected Edition (Smith, 1776, Book 1, Chapter 4), where he describes the paradox of value between water and a diamond: "The word Value … has two different meanings, and sometimes expresses the utility of some particular object, and sometimes the power of purchasing other goods which the possession of that object conveys. The one may be called 'value in use'; the other 'value in exchange'. The things which have the greatest value in use have frequently little or no value in exchange; and, on the contrary, those which have the greatest value in exchange have frequently little or no value in use. Nothing is more useful than water; but it will purchase scarce anything; scarce anything can be had in exchange for it. A diamond, on the contrary, has scarce any value in use; but a very great quantity of other goods may frequently be had in exchange for it."

Although water in modern times has become a much scarcer commodity than in Adam Smith's time, it is still true that water is a relatively inexpensive commodity. However, it is also a social good, and the rights to free basic water and to an ecological water reserve are recognized in South African water law.

It is likely for this reason that the Pricing Strategy takes a cautious approach and links economic charges implicitly to water use licenses, whilst using compulsory licensing as a pre-condition. However, the Pricing Strategy does not express an explicit opinion on this apparent conflict.

Revenue potential of economic charges

DWS, during its National Water Policy Review in 2013, indicated an intention to strongly regulate water transfers. Although the NWA makes provision for transfer of water use authorizations (Section 25), since 1998, instances of water trading have emerged that have undesirable unintended consequences such as, for example, that the water allocation may take place without Ministerial consent; and that water, because it is a scarce social good, becomes commoditised, and water traders use it as a means to make profit, at the cost of users. Thus, although DWS has signalled policy intent to prohibit water trading, economic charges still remain a valuable policy instrument in water transfer mechanisms.

The framework provided by the current Pricing Strategy gives insufficient policy direction for economic charges.

3.2.3.2 Bulk Water Tariff

Prices for bulk water provided by water boards are set by water boards themselves, subject to Ministerial approval. Where WSAs provide bulk water to other WSPs, price and other terms are negotiated between the parties. It recovers the cost of conveying and treating bulk water and wastewater. Water boards who undertake social infrastructure receive funding from Regional bulk infrastructure transfer through DWS and despite this grant, more than half of water boards cannot recover their costs.

3.2.3.3 Municipal Water Services Tariff

Municipalities especially in the rural areas are subsidised by government through a number of grants including: The Water Service Infrastructure Grant (WSIG); DORA (Division of revenue act); Municipal Water Infrastructure Grant (MWIG), etc. The government through the grants helps municipalities to develop new, as well as refurbish, upgrade and replace ageing infrastructure that connects water resources to households across municipal boundaries. Despite all these grants as mentioned above, only one metropolitan municipality can recover its costs.

There are a number of obstacles to the implementation of proper ring-fencing and these include the lack of cost accounting and the highly technical nature of water service provision that make intra-departmental communication difficult. Furthermore, many municipalities are forced by poor financials to cross subsidise services to manage cash flow or to fund other non-revenue services. This results in ring-fencing revenue collected for maintenance and upgrading of WWTWs difficult to achieve.

3.2.4 Complying with legislative requirements

3.2.4.1 Compliance at National Level

The DWS describes itself as “sector leader and national water policy maker” (DWAF, 2003). It describes its functions as “enforcer, enabler and supporter” (DWAF, 2005). These multiple roles are not always reconcilable. As far as the value chain is concerned DWS administers regulations on tariffs promulgated in terms of section 10 of the Water Services Act. It routinely monitors water services tariffs and publishes them annually. Lindfors (2011) describes, the DWS has the overall responsibility of water resource management and water service provision. As the leader of the water sector, its role is to support and strengthen the WSAs.

The DWS's (2013) review of the water sector in the country indicated that *DWS oversees and regulates the water business through appropriate policies and regulations which are implemented through its 9 provincial offices and 4 water management clusters. DWS also monitors the performance of the sector and regulates the drinking water quality and effluent quality against industry standards and recommends changes to the business environment within which the various role players have to perform*.

DWS (2013) further indicated that *the National Water Services Regulation Strategy has been adopted and the Blue and Green Drop incentive based regulatory processes have been conceived. These have been implemented with a visible improvement in drinking water and wastewater quality. Much had been done to assist local government to effectively deliver on its mandate.* The monitoring sector performance (including conformity to national norms and standards) and monitoring compliance with (and relative performance with respect to) national policies was addressed through the Blue Drop; Green Drop and RPMS performance systems which were introduced to the water services from 2008.

Transparency in regulation is also a challenge, despite the water and sanitation sector being extensively regulated by legislation, regulations, standards, policy and guidelines, there is no independent regulator. The DWS assumed complex and often conflicting roles of regulator, supporter, enabler and implementer in the water resources management arena. As a consequence, enforcement remains weak.

3.2.4.2 Compliance at Local Level

The SFWS indicated that water services authorities might regulate the provision of water services within their local area through these by-laws and contracts (DWS, 2003). The by-laws set out the general rights, duties and responsibilities of water services providers, intermediaries, water services agents and consumers with respect to water services. The relevant by-laws and consumer contract must set out or provide for consequences and remedies when a consumer breaks a by-law or fails to meet the contractual obligations with the water services provider. Council approval of water by-laws is the endpoint of a long process of development, review and refinement of by-laws (DWS, 2003).

Interviewees in this scoping study indicated a lack of implementation of bylaws by relevant authorities.

3.2.4.3 Political Landscape

Political frameworks preserve political discretion on key issues such as tariffs and prioritize mechanisms for consultation with labour and consumers over the structure of water services (Morgan, 2006). However, in the South African water sector particularly the municipal context the process of prioritising projects under the Integrated Development Plan (IDP) consultations is subject to some degree of political interference. Political motives tend to be short-term, whereas water resource management requires long-term strategic thinking.

The 1994 White Paper of Water Supply and Sanitation Policy indicated that an important function of the Central Government is, whilst devolving implementation of water supply and sanitation to the lowest level possible, the monitoring, auditing and reporting of the sector (DWAF, 1994). The policy in this White Paper is aimed at opening up the arena for as many participants in development of basic water supply and sanitation services, which would necessitate the need to monitor and regulate their performance (DWAF, 1994).

3.2.5 Consumer Behaviour

Domestic water use has a large share in water and could largely be attributed to household activities such as drinking, food preparation, bathing, washing and toilet flushing (Hollingworth, 2011) and especially in consumptive water use for watering of gardens and similar.

A recurring theme of various water policies in South Africa is cost recovery and the associated "user pays" principle (South Africa, 1994). The NWRS is very explicit on this matter, envisages that economic regulation of water is a neglected area in the South African context. It acknowledges a greater need for using water pricing mechanisms as a policy instrument for Water Conservation and Demand Management (WCDM), and it indicates explicitly that "the price of water does not send the correct economic signal that water is a scarce resource". Several interviewees have echoed this sentiment and believe that smarter block tariffs are required as a WCDM mechanism.

Municipalities are faced with the challenge that the end-users, particularly domestic users, who are unwilling to pay, find ways to by-pass the meters and are therefore not paying for their water use. WSAs therefore have to find ways to isolate water meters to prevent tampering and diminishing the impacts of "unaccounted for water", but municipalities also need to ensure that they bill users correctly and collect the revenue due to them.

Due to similar challenges in unaccounted water across the country, the DWS had introduced the No Drop performance management system. This system reports the levels of non-revenue water of a WSA, which is effectively the amount of water supplied in the WSA for which no payment is received. Although a portion of this water is related to the free basic water component of water supply, non-revenue water could have a significant impact on the financial sustainability and the functioning of a WSA and thus their ability to ensure water security within their jurisdiction.

There is need for a suasion and education initiative to make the public aware of water conservation and understanding of the process of the water value chain.

3.2.6 Roles and Responsibilities

There seems to be confusion on roles and responsibilities between DWS, CMAs, bulk water suppliers and WSAs. For example, Umgeni Water used their bulk water charge levy to clear alien invasive species and they were told it is not their responsibility but that of a CMA.

Retail charge through WSAs has an ecological infrastructure charge where the service provider has the opportunity to fix wetlands, but the complexity is with working outside municipal boundaries

3.3 Overview of Key Challenges Identified

South Africa has different instruments of internalising ecological degradation and is mainly through raw water pricing. This is due to the fact that regulatory approaches have generally been preferred to date and have in some cases been relatively successful in reducing pollution.

One of the contributors of point source pollution is WWTWs, even though the DWS and COGTA try to find ways to minimise this problem through Green Drop certification and several grants for poor municipalities. The challenge is that DWS is the enforcer and the supporter to municipalities and these conflicting roles makes enforcement very weak.

With regards to financial constraints, non-payment and historic debt were major challenges for DWS and WSAs. This results in Wastewater Treatment Works (WWTW) not being maintained and repaired.

Current regulatory policy instruments cannot hold the private sector to account for any damages caused by AMD if the owners of the mines do not exist anymore and AMD is largely caused by defunct mines which are ownerless and have no legal provision.

Regulatory policies require heavy monitoring of polluters and currently DWS does not have the capacity to administer that at all times. This lack of resources leads to weak enforcement.

It is interesting to note that none of the interviewees identified technical challenges but rather implementation challenges. Therefore, EPIs to be proposed in the section below must be able to be implemented with minimum administration required.

4 AN EPI FRAMEWORK

4.1 Components of an EPI Framework

The point of departure for identifying EPIs is the Millennium Ecosystems Assessment (MEA) ecosystem services framework. This is because aquatic ecosystem services define the linkages between water resources and the economy, and EPIs seek to institutionalise these linkages into economic transactions.

In addition to ecosystem service transaction mapping, a regulatory review was conducted, international literature review and expert interviews, to further develop a conceptual framework for EPIs. The resultant framework takes the form of a matrix and is presented in Table 4-1.

This work resulted in a conceptual framework that characterised EPIs on the basis of five elements. These five elements include:

- **Ecosystem Service Category and User / Impactor** – The Ecosystem Service Category defines the broad category of ecosystem services. The User / Impactor defines the economic actor either using natural resources or who are responsible for ecological damage/degradation (through pollution or other impacts), i.e. the party who would be responsible for payment.
- **Regulator / Implementer** – These are the role players required to ensure EPIs are appropriately institutionalised. They are also the organisations who would be party to the transactions required to make EPIs work. It includes regulators who would have market oversight roles, Users / Impactors (also refer to 5 below, some Beneficiaries may also be transaction parties).
- **Transaction clearing mechanisms and institutional arrangements** – These are the institutional arrangements required for clearing transactions within the EPIs. Clearing here is defined in the financial market sense, i.e. all the activities required from the moment a commitment is made to the point of settlement. Refer to section 4.2 for a more comprehensive discussion of clearing mechanisms.
- **Specific benefits provided** – These are the detailed ecosystem services benefits provided, with chains of causality specified according to the MEA and TEEB ecosystem services framework. These chains of causality would define the ecosystem services production process and would therefore also inform the evidence-base for EPI transactions, as well as the transaction monitoring requirements.
- **Beneficiaries** – These are the beneficiaries of the EPIs. It is expected that there would be two categories of beneficiaries, those that benefit directly from EPI transactions, and could therefore be party to EPI transactions; and those that benefit indirectly through public good benefits.

In order to demonstrate some of the salient features of EPIs, key challenges inherent to making market-based instruments work, need to be considered. These are typical transaction feasibility considerations. Refer to section 4.3 below for a discussion of these.

In addition, we have selected four types of EPIs for further investigation. This selection was done to demonstrate a number of characteristics of EPIs. These include a water tariff system, industrial waste-water charge with a deposit-refund system, conservation credit trading and the WCDS. The analyses performed on these EPIs are presented in section 5 below.

Table 4-1: Preliminary summary of proposed EPI, beneficiaries, and the benefits they provide

Ecosystem Services Category	User / Impactor	Regulator / Implementer	Potential EPI (Proposed)	Transaction clearing mechanisms and arrangements	Ecosystem Services / Benefits	Beneficiaries
Water use	Raw water, bulk water and water services users	DWS, WSAs / DWS, CMAs, WUAs, Water Boards, WSAs	Basket of policy relevant (equity, efficiency, sustainability) water charges	Water tariffs	Water regulation and water provisioning / Equitable, sustainable and efficient (cost recovery and scarcity geared) water use across the value chain	Water users along the water value chain
Habitat provisioning and ecological infrastructure	Cumulative degradation	DWS, WSAs / DWS, WSAs	Green infrastructure management	Commoditisation, certification	Environmental asset improvement and its associated ecosystem services	Beneficiaries of relevant and affected ecosystem services
	Mining; Construction; Urban developers; Agriculture	DWS, DEA, DMR, WSAs / CMA, WSAs, Industry	Land and wetland rehabilitation	Tradable permits; certification	Improved water quality and its associated benefits; increased water yield; environmental asset improvement and its associated benefits	Raw water users (Water Boards, Irrigators, Eskom, mining industry); Beneficiaries of relevant ecosystem services
Water purification and waste assimilation	WWTWs; Industries (e.g. mining, manufacturing)	DWS/CMA	WDCS	Water tariff, environmental tax	Improved water quality downstream, and its associated benefits	Water purification and waste assimilation / Raw water users (Water Boards, Irrigators, Eskom, mining industry)
		WSAs/WSAs	Industrial wastewater charge	Water tariff	Improved water quality downstream	
		DWS/Industry	Polluted water treatment scheme	Deposit refund system; certification; environmental auditing	Improved water quality downstream; increased water yield	
	Agriculture runoff (polluted)	DWS / CMA, Agriculture	Water pollution offsets	Tradable permits	Improved water quality	Raw water users (Water Boards, Irrigators, Eskom, mining industry)

4.2 About Transaction Clearing Mechanisms and Institutional Arrangements

4.2.1 Definitions

For EPIs to be effectively implemented, they require a range of transaction clearing mechanisms, i.e. the mechanisms that make markets work. These mechanisms can be thought of as institutional arrangements that enable transactions to be cleared. In other words, they comprise the range of institutional arrangements that enable the completion of all activities required, from the time a commitment in a transaction is made, to the point where the transaction is settled. This includes market oversight functions. Transaction clearing mechanisms could include protocols, systems, organisations and trading platforms.

The sections below elaborate on these arrangements.

4.2.2 Tariff[2] Systems

A well-functioning tariff system is a key transaction clearing mechanism underlying the successful implementation of EPIs.

“Well-functioning” in this case means firstly that water tariffs or charges are set accurately through a regular and appropriate review process. Accuracy here refers to the specific policy objectives, e.g. if the policy objective is full cost recovery, the tariff needs to be cost-reflective. Secondly, it also means the tariffs are policy relevant and therefore effective. This would include policy effectiveness related to the equity, efficiency and sustainability imperatives. Thirdly, the tariffs need to be implemented through an effective invoicing and payment system. Finally, the revenue collected needs to be spent effectively.

South Africa has, conceptually, a well-developed water tariff system framework, comprising policies, norms and standards, and review processes across the value chain. At the heart of water South Africa’s water tariffs have been designed and implemented to promote equity, for example, through the free basic water approach. Many municipalities pursue efficiency through rising block tariff strategies. The Raw Water Pricing Strategy contains a tariff component reserved for catchment management. The WDCS system is envisaged to address an aspect of sustainability.

[2] The term “pricing” is widely used with the common definition of a per unit value for water at a certain quality. It is to be noted that a “price” strictly speaking, is only relevant where a well-functioning market of buyers and sellers exist, and where the resultant price is a reflection of the relative scarcity of the commodity traded. It is therefore more correct in the South African water management setting, to refer to “water charges” or “administered tariffs”.

However, the tariff system is arguably not "well-functioning". Refer to section 5.2 below for a more detailed discussion on this.

4.2.3 Environmental Tax Systems

In literature, "environmental tax" is loosely referred to as an EPI. We make a distinction between the concept of an "environmental tax system" and an environmental tax-based EPI. An environmental tax system comprises the necessary institutional arrangements, in South Africa governed by National Treasury, to design and implement a specific environmental tax-based EPI.

This distinction is important because, on its own, an environmental tax is not a market-based EPI, but rather a fiscal policy instrument (refer to Table 2-2).

In general, tax is commonly classified according to its base and includes the following categories:

- Income, profits and capital gains taxes
- Payroll taxes
- Property taxes
- Goods and services taxes
- Other taxes.

Environmental taxes fall within the category of "other taxes".

Environmental taxes are a fairly well-developed concept, and Pigou (1920) was the first to describe the environmental taxation mechanism. Accordingly, increasing negative environmental externalities from economic development should proportionally multiply the sum of environmental taxes.

Fundamentally, an effective environmental tax is expected to generate sufficient revenue to (a) internalize (i.e. avoid) environmental damages and (b) capture relevant resource rents related to natural resource use and ecosystem services benefits into the economy. In addition, an effective environmental tax will result in fiscal benefits and in national welfare gains.

The practical implications of Pigouvian tax are well summarised by the European Environmental Agency (EEA, 1996):

- Firstly, they are instruments for the internalisation of environmental externalities in the economy, i.e. the incorporation of the costs of environmental damage into the prices of the goods, services or activities which cause them. (This is therefore associated with the implementation of the Polluter Pays Principle.)

- Secondly, they provide incentives for both consumers and producers to change their behaviour towards a more eco-efficient use of resources; to stimulate innovation and structural changes; and to reinforce compliance with regulations.

- Thirdly, they can raise revenue which may be used to mitigate environmental damage.

The EEA mentions in particular that environmental taxes can be policy instruments through which to address diffuse pollution sources such as transport emissions and waste.

The EEA classifies three main types of environmental taxes, according to their main policy objectives. These are:

1. **Cost-covering charges** – e.g. designed to cover the costs of environmental damage and abatement measures (such as water treatment)

2. **Incentive taxes** – designed to change the behaviour of producers and/or consumers

3. **Fiscal environmental taxes** – designed to raise revenues (i.e. capture resource rents).

Because environmental taxes are not only revenue raising instruments but are also part of a set of environmental policy tools that seek to internalize environmental externalities, the IMF (1998) adopted the term "double-dividend". The first dividend is an improvement in the environment, and the second dividend is an improvement in economic efficiency from the use of environmental tax revenues to reduce other taxes such as income taxes that distort labour supply and saving decisions.

An effective, double-dividend, environmental tax system also enables tax reductions to be used as incentives. Rather than charging an impactor for emissions, impactors are financially rewarded if the levels of pollution are reduced to the acceptable range. Tax reductions have been used for a wide variety of purposes, including: agricultural grants for erosion control; low-interest loans for small farmers; grants for land conservation; and loans and grants for recycling industrial, commercial and residential products. While this offers incentives to reduce emissions, it also encourages market entry to qualify for the subsidy (Bakker, 2009). Nonetheless, there are some disadvantages to using them and they include:

- People might object to the government paying the responsible companies to reduce their pollution.
- Companies could become dependent on the financial benefit of the tax reductions, so much so that their business model might hinge on getting the subsidy. As a result, incentives can pose a drain on government resources (O'Connor, 1994).

4.2.4 Tradable Permit Systems

Tradable permit systems are so-called cap and trade schemes, which involves rights to sell and buy actual or potential endowments (e.g. emission allowances) in artificially created markets. A trading scheme can therefore be defined as a regulated framework which establishes certain assets and commodities, according to specific market-based regulations, generally related to demand-supply mechanisms.

Arguably the best-know trading scheme is the European Union's Trading System (ETS) for trading greenhouse gas emission allowances. This scheme sets a cap on the total emission allowed, with allowances adding up to the cap provided to the companies regulated by the scheme. Within the cap, companies receive or buy emission allowances which they can trade with one another as needed. The limit on the total number of allowances available ensures that they have a value. After each year a company must surrender enough allowances to cover all its emissions, otherwise heavy fines are imposed. If a company reduces its emissions, it can keep the spare allowances to cover its future needs or else sell them to another company that is short of allowances (Bertoldi and Huld, 2006).

The advantage of tradable permits is that polluting organisations differ in their ability to abate their pollution. Some can do it easily and cheaply, for others it would be more difficult and costly. Consequently, tradable pollution permits can be a cost-effective way to achieve a reduction in overall pollution.

The two main ways of initially allocating tradeable pollution rights are usually referred to as ''grandfathering'' and ''auctioning''. Grandfathering involves allocating permits to companies on the basis of their past emissions. Companies that polluted more in the past would have larger shares. Alternatively, a prespecified number of pollution allowances can be auctioned off to impactors. In either case the total allocation can be based on the estimated capacity of the environment to take a certain amount of pollution. If the permit that a company is allocated or has bought is less than their actual emissions, then such companies would have to either try and reduce their emissions or buy extra permits. Similarly, they would be able to sell those they don't need if they reduce their emissions below what they are permitted to emit.

Grandfathering favours existing companies and disadvantages new companies wanting to set up. In order to establish itself, a new company must buy up enough pollution rights to cover its emissions. Ironically, it is often easier and cheaper to install clean technology processes when a company is newly established than to repair an older established company that has outdated and polluting equipment. Alternatively, the government can increase the amount of rights available and give the new company an allocation. This latter option will increase the amount of pollution and defeat the purpose of trying to reduce overall emissions (Stavins, 1997).

Auctioning means that each company has to bear additional costs just to operate as they have to buy permits at auction to emit pollution they had previously been emitting for nothing. This is especially hard for companies that are competing with overseas companies not having to bear these costs. It is for these reasons that auctioning appears to be less acceptable to industry, than grandfathering.

A major drawback of trading is that it can cause some neighbourhoods to get a lot more pollution than others because the companies in their area are buying up permits rather than reducing their pollution. Also, trading tends to protect very polluting or dirty industries by allowing them to buy emission rights rather than meet environmental standards. In this way, trading can reduce the pressure on companies to reduce their emissions, for example by changing their production processes (Stavins, 1997).

It is therefore important to have tradable permit systems that are effective and that allow for suitable market oversight.

4.2.5 Deposit Refund Systems

The best-known example of a deposit-refund system in South Africa is the system of fees and reimbursements for beverage containers. These so-called ''bottle bills'' were adopted to combat litter problems and have been operating successfully. This approach can also be used to address many other environmental problems well beyond waste disposal, such as water pollution. By imposing an up-front fee on production or consumption and using those fee revenues to rebate ''green'' inputs and mitigation activities, a deposit-refund policy may be able to efficiently control pollution in much the same way as a Pigouvian tax (Fullerton et al., 2008).

A large portion of public environmental expenditures is for restoration of degraded environments, which could have been prevented or paid for by the impactors or beneficiaries of responsible activities and as a result, the deposit refund system shifts responsibility for controlling pollution, monitoring, and enforcement to individual producers and consumers who are charged in advance for the potential damage. A great advantage of deposit-refund systems is the inducement of a labour-intensive activity in an environment of low-cost, abundant, and underemployed labour.

Induced self-regulation is more efficient and cost effective than direct government regulation because industries know best how to control their own waste, because self-enforcement is induced by a desire to be accepted by other members of the association and by the community, and because the cost of policing and monitoring is significantly reduced and assumed directly by the source. Again, the funds needed for environmental clean-up and enforcement of environmental regulations are reduced and generated from among the members of the industrial association in a manner that alters behaviour and way of doing business.

A deposit refund system is considered to be more cost-effective than other methods, but the relatively high administration costs of a deposit system could outweigh these cost savings.

4.2.6 Certification Systems

Certification is one of the available tools in the market to ensure the application of principles for sustainable production of commodities. It comprises a set of principles addressing social and economic concerns. Different certification schemes vary in their main focus or strategy for achieving a more sustainable production with some of them focusing on the creation of sustainable trade relations.

Successful certification systems are premised on a scientifically-founded and market-based set of indicators. These indicators enable measurement and auditing of value chain practices against criteria such as best conservation practices, fair and ethical business practices and equitable trade practices. These indicators are captured within a formal standard, accompanied by implementation guidelines.

Various institutional models for certification exist. In highly organised and vertically integrated value chains (e.g. typical of the wine industry in South Africa), certification systems are fully implemented through industry organisations. In complex value chains such as cocoa production, forestry and bioprospecting, however, certification systems are rather administered by reputable NGOs or independent and specialised private companies. These latter models require investment in branding, support and auditing infrastructure that create incentives for role players in the value chain to adopt one or more certification systems. Global examples of such certification systems includes the UEBT-GIZ, SECOSA, Fairtrade, Forest Alliance, UTZ, Certified, Forest Stewardship Council, COSMOS, Freewild and others.

Product labelling is one form of certification, which involves certifying products that are produced in a way that is environmentally preferable to other products in the same product/service category based on life cycle considerations. Product labelling is meant to create consumer preference for “green” products and thus generate a financial return to the supplier of the certified product in the form of increased revenues. Labelling of agricultural products can provide incentives for farmers who wish to certify their products and adopt sustainable agricultural practices.

A study in Ivory Coast on cocoa production (KPMG, 2012), found that certified farmers appear to be somewhat more community-minded than their non-certified counterparts, demonstrating a higher rate of participation in community projects, such as building community agricultural facilities, roads or schools.

In South Africa, during the development of the DEA National Biodiversity Economy Strategy in 2016, the potential of certification systems specifically for the wildlife economy and bioprospecting sectors was identified. The DWS’ Blue and Green Drop certification systems have in the past been implemented.

Such systems may form the basis for market-based EPIs and need to be further explored in innovative ways.

4.2.7 Assigning Property Rights

An interesting phenomenon arising from the concept of ecological infrastructure is the emergence of ecological commodity properties of water resources. For instance, an early example of this is the Wetland Mitigation Banking system in the US (Robertson, 2004). The Clean Water Act gave the Corps of Engineers the power to issue developers with permits to allow the damage of wetlands in exchange for their commitment to create or restore equivalent area wetlands elsewhere. From this system, it turned out that the average cost of wetland mitigation was approximately US$45 000 an acre, putting in practice a market price on preserved wetlands (Bayon, 2004).

In economics, a commodity can be defined as a basic good used in commerce that is interchangeable with other commodities of the same type. Commodities are most often used as inputs in the production of other goods and services. The quality of a given commodity may differ slightly, but it is essentially uniform across producers. Commodities often show the following characteristics:

- Clear demand and supply for a natural resource
- Limited geographic distribution and supply
- Global or regional market depends on it for providing benefits;
- Uniformity and fungibility – commodities are graded and classified as a specific type so that their characteristics are fairly uniform
- Has a per unit value.

The perception of "commodification" of the natural environment has received criticism (Kosoy and Corbera, 2010). Such criticism is based on:

- An ethical consideration which regards ecological assets as fundamental to life and therefore invaluable; and
- A view that environmental assets are public resources, which should not be traded for profit.

The response to this criticism lies in the concept of substitution. Substitution is the process of purchasing one commodity in place of another. This definition, however, fails to account for commodities that are not perfectly substitutable. Many commodities are closely related; yet in the eyes of the consumer not perfect substitutes. It is useful to think of ecological commodities as lying on a continuum according to how closely they are related. At one extreme are perfect substitutes, at the other perfect complements. In the middle are independent commodity pairs. In between either extreme is a range of closely related ecological commodities with some degree of substitutability.

There may therefore exist cases where some wetlands are not substitutable, i.e. become no-go areas. In other instances, wetlands may be substitutable with other wetlands. The degree of substitutability and the cost of substitution are not primarily determined in a market, but through scientific and engineering assessments of the costs entailed in restoration and ongoing maintenance of a wetland.

An example of an EPI that internalises wetland asset value into the economy would be to incorporate the intrinsic value of wetland ecosystem services, as the discounted value of perpetual wetland ecosystem services, into land value. Precedents for this already exist in some urban environments where aesthetic benefits of natural environments increase property value and further increase municipal tax revenues.

Various mechanisms exist through which property rights may be granted. Obvious and common examples include conventional tenure systems that grant access to land (outright ownership, general authorisations that grant access, and servitudes), concessions and auctions.

4.2.8 Environmental Auditing

Environmental auditing is an important mechanism for enabling effective market oversight. South Africa is arguably highly dependent on environmental regulators (e.g. DWS and DEA) for monitoring and enforcing environmental compliance. An effective environmental auditing sector would significantly reduce this burden and reduce transaction costs.

Environmental auditors would monitor, verify and report (i.e. audit) whether an organisation is operating in an environmentally responsible manner, i.e. that complies with the organisation's environmental policies and environmental legislation. In addition, environmental audits identify weaknesses and risk areas in terms of potential environmental disasters and provide a forum for the exchange of technical knowledge and the identification of areas where costs can be saved (Edwards et al., 1992; LEAF, 1994)

While environmental audits are designed to identify environmental problems, there may be widely differing reasons for undertaking them: compliance with legislation, pressure from suppliers and customers, requirements from insurers or for capital projects, to demonstrate environmental activities to the public, or to audit compliance to certification systems (refer to 4.2.6).

4.2.9 Valuation Systems

It is expected that a variety of valuation mechanisms need to be instituted to support the development of EPIs. These mechanisms would be useful during the initial design of the EPIs, especially in the case of issuance of permits, arrangement of auctions, determination of tax levels for double-dividend objectives and similar mechanisms. These valuation systems are likely to comprise mostly resource economic tools. However, tools such as the DWS wetland offset calculator, which estimates equivalent hectares for wetland offsets based on relative scarcity, would also be defined as a form of valuation tool (SANBI & DWS, 2016).

Valuation systems need to provide consistent, transparent and reliable outputs.

4.2.10 Regulatory Policy Instruments and Regulatory Organisations

These instruments form a fundamental part of the institutional arrangements required. These instruments would inform, for example, thresholds to be adhered to, caps that may be set, numbers of permits that may be issued, or tax levels that need to be set (where an environmental tax forms part of an EPI).

Regulatory organisations are also expected to play active market oversight roles. This could include specific roles to be played in monitoring whether policy goals are being met, reviewing environmental auditors' reports, issuing permits, setting conditions, issuing sanctions for non-compliance and other roles.

4.2.11 Criteria in Designing Appropriate Economic Policy Instruments

Feasible EPIs should include the following criteria (adapted from Bernstein, 1997):

1. Designed with the following parameters in mind:
    a. Users / impactors need to be clearly identified
    b. Benefits and beneficiaries need to be clearly identified
    c. Ecosystem Service to be internalised and industry to benefit.

2. The policy objective of the EPI needs to be clearly defined, and EPIs should be designed in combination with regulatory measures. Equity considerations should be carefully balanced with environmental factors when selecting instruments. A major policy question when considering any tax system is who, ultimately, will bear the burden of the tax? Or, does the tax fall proportionately more on the rich or the poor?

3. An appropriate combination of transaction clearing mechanisms (refer to section 4.2) need to be applied to ensure the EPI is effective. The implementation role players and mechanisms need to be clearly described.

4. A fatal flaw analysis needs to be conducted.

5. It is crucial to respect transparent, democratic and bureaucratically feasible processes for decision making. The parties affected must be given the opportunity to influence it for the sake of legitimacy and because they are the best sources of information. On the other hand, the

parties can obviously not be given too much influence if this means that effective instruments are ruled out. To understand the politics of policy design, careful attention must be paid not only to the way the instruments work but also to the distribution of costs that they imply, between the impactors and others in society.

6. The flexibility of the instrument in adapting to a changing environment can be an important consideration where there are changing local conditions. For example, depending on local political conditions, changing a charge rate may be more easily accomplished than changing legislation, except of course if the rates are set within the legislation. Environmental taxes also confer, on producers and consumers, the flexibility needed to minimise the costs of achieving a given goal. To the extent that different organisations can have different costs for pollution abatement, a charge can encourage those facing lower abatement costs to go further in cleaning up their operations.

7. An instrument should be selected only if the responsible agencies are prepared to deal with the often-complex procedures required for implementing them properly, such as billing and collecting taxes and charges, measuring emissions, determining environmental effects, and taking the necessary enforcement action for non-compliance. All of these require good co-ordination between government departments and implementers.

8. In selecting an instrument, it is important to select for an instrument that can achieve the desired outcome at the least possible cost and with a total cost that does not exceed the expected benefits. The optimal instrument is one that leads to the so called "win-win" solutions, i.e. improvements in the environment and other sectors of the economy occur simultaneously and therefore do not involve difficult development-environment trade-offs.

These criteria are considered in section 5 below, in the demonstration of four selected EPIs.

4.3 Potential EPIs for the Water Sector in South Africa

Following from Table 4-1, we define seven possible EPIs. This is not intended as an exhaustive list, and it is possible that additional, innovative EPIs could be conceptualised.

1. The DWS Raw Water Pricing Strategy envisages an “economic charge for water” that would be applicable to water rights for productive use post compulsory licencing. This is envisaged to be implemented through either administratively determining a proxy for the economic value of water, or by selling certain water rights by public tender or auction to the highest bidder. The economic charge is envisaged to promote beneficial use through the reallocation of water to higher value users. Where amounts of water are still available for allocation after compulsory licenses have been issued and there is competition for using this water, the public tender procedure may also be followed.

2. The DWS Raw Water Pricing Strategy also envisages a Waste Discharge Charge System (WDCS), which is a “polluter pays” system. The WDCS consists of two distinct water use charges: an incentive charge that provides a disincentive to the discharge of waste; and a mitigation charge that covers the quantifiable costs of administratively implemented measures for the mitigation of waste discharge related impacts.
3. Several municipal by-laws envisage industrial wastewater charges, a special category of waste-water charges that focus on minimising particular water emissions.
4. Water pollution permit trading could be an alternative to a WDCS and adopt the cap and trade system used in mitigating air pollutants.
5. Green infrastructure management systems could be used to internalise ecosystem assets into existing State-operated immovable asset management systems. This would require these assets to be registered, valued and managed within existing budgetary processes.
6. Eco-restoration permit trading is an EPI that could follow from conditions associated with environmental authorisation processes, e.g. wetland offset requirements.
7. Pollution deposit-refund systems are systems where impactors may purchase pollution concessions and then get refunded for reducing emissions.

5 DEMONSTRATION OF SELECTED EPIS

5.1 Overview

Table 4-1 demonstrates a set of permutations of EPI applications. The Table identifies users / impactors, implementers, EPIs, transaction clearing mechanisms and beneficiaries. The key water users were identified as irrigation farmers; industry (mining, manufacturing); WWTW; developers (i.e. construction). Policy instruments were identified that could mitigate these impacts (Table 4-1). However, based on the South African landscape, we examine the following EPIs that have the potential to internalise EEB in pricing of water and its subsequent management: These EPIs is a subset of the seven EPIs listed in section 4.3 and they demonstrate a variety of characteristics of EPIs.

1. **Water tariff**- This EPI analyses tariffs for water resource management across the water value chain, i.e. DWS; water boards; and municipalities.

2. **Industrial waste water charge with deposit refund**- The EPI assumes municipalities that implement the industrial wastewater charge will incentivise change in the behaviour of polluters by setting maximum thresholds on emissions of specific harmful pollutants. This will be done by overcharging the industries. Industries that emitted below the set water quality standards will be refunded the difference. Using deposit refund, industries will co-operate more and as a result, change their behaviour

3. **Conservation credit trading**- The bank sponsor will be responsible for success of this EPI. The usefulness of this trading is greater where the demand to develop wetlands is high. Conservation credit trading reduces uncertainty over whether the compensatory mitigation will be successful in offsetting project impacts.

4. **WDCS with tradable permits**- WDCS will be implemented on a catchment level and CMAs will be able to raise revenue from WWTWs, mines and farmers. Tradable permits will be allocated amongst users, and those who cannot emit below the standards can buy from those that emit below the standards. The success of this EPI will result in reduction of contaminants in water resources and this will mainly benefit water services providers and farmers.

Water tariff and industrial wastewater charge will be demonstrated in a narrative manner and conservation credit trading and WDCS will be demonstrated through models that are briefly discussed below.

We have adopted the pricing of investments assets approach to the procedures of pricing EPIs. This approach assumes that the mark-up value (the profit taken by seller) is solely to the preference of the seller and the negotiation between the seller and the buyer. This approach of pricing an instrument has two aims. The first aim is to price the EPI in order to recover the costs incurred by the authorities. The second aim is to price the price effect of demand and supply in the instrument in such a way that the instruments' scarcity is reflected in the price. The following sections briefly discuss how the WDCS with tradable permits and conservation banking instruments are priced.

5.2 Water Tariff for Ecological Damage Mitigation

The Water Pricing Strategy makes provision for a water tariff component for sustainable water use, in the form of the WRMC. This charge comprises a catchment management charge, within the raw water pricing tier.

For the purposes of this discussion, it is important to first define sustainable water use.

Sustainable water is defined by adopting the FAO (2004) analytical framework for economic water charges. This concept is entitled the "full cost of water consumption". According to this definition, the full cost of water consumption comprises three cost components: the Full supply cost, the Full economic cost, and the Full cost (Rogers at al., 1998). The sections below were adapted from Rogers et al. (1998).

Figure 5-1 provides a schematic representation of these charges.

The **Full supply cost** is defined to include the costs associated with the supply of water to a consumer excluding the externalities imposed upon others and excluding the alternate uses of the water (i.e. opportunity costs). The Full supply cost comprises two items: Operation and Maintenance (O&M) Cost, and Capital Charges, both of which are defined to include the full economic cost of inputs. O&M costs are associated with the running of the supply system. Typical cost items include purchased water, electricity for pumping, labour, repair materials, and input costs for managing and operating storage, distribution, and treatment plants where relevant. Capital Charges include capital consumption (or depreciation charges) and interest costs associated with reservoirs, treatment plants, conveyance and distribution systems.

The **Full economic cost** of water is the sum of the Full supply cost, the Opportunity cost associated with the alternate use of the same water resource, and the Economic Externalities imposed upon others due to the consumption of water by a specific actor. The Opportunity cost addresses the fact that by consuming water, the user is depriving another user of the water. If that other user has a higher output production value associated with water use, then there are some opportunity costs experienced by society. The Opportunity cost of water is zero only when there is no alternative use (or where there is no shortage of water). The consequence of ignoring the Opportunity cost undervalues water, and results in a failure to invest optimally. Economic externalities relate to over-abstraction of water. This may include water losses (unaccounted for water), over-extraction from common pool resources such as underground water, and wasteful use of water.

The **Full cost** of consumption of water is the Full economic cost, as defined above, plus the Environmental externalities. Environmental externalities are those associated with water resources management or maintenance of ecological infrastructure and water pollution.

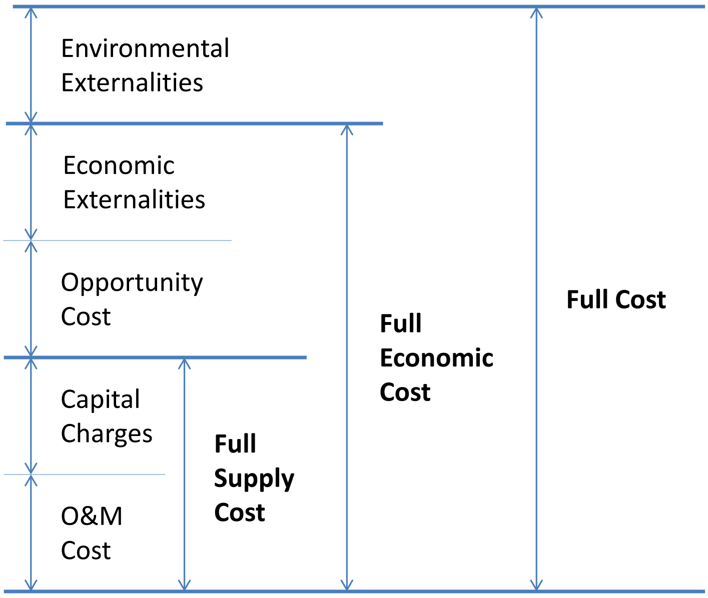

Figure 5-1. The Full Cost of water consumption comprises the components defined in this figure. This includes O&M Cost, Capital Charges, Opportunity Cost, Economic Externalities and Environmental Externalities (Rogers at al., 1998).

Water tariffs promoting sustainable use fall in the Economic externality and Environmental externality components.

The question now arises how the environmental externality is defined. An earlier WRC study (Pearce et al., 2014) proposed that to fully address environmental externalities, a tariff structure has to:

- Comprehensively address water resource management, across all three water pricing tiers
- Address water losses
- Address over-abstraction of water.

Through WRMC, South Africa has an existing policy instrument through which water resource management in Tier 1 is addressed.

The following table is a direct extract from the raw water pricing strategy (DWA, 2007). It contains four key considerations pertaining to the allocation of water resource management charges:

Sectoral water resource management charges for each WMA will be calculated as follows: 1. Total budget cost of each activity will be divided by the registered volumes to arrive at a unit charge per activity. 2. In WMA's where the allocable yield (water available for use) is more than the registered volumes, a discount will apply, which will be determined by using the allocable yield to determine the per unit charge instead of the registered volumes. 3. The budgeted activity cost will be applied only to those sectors attracting such cost (e.g. the forestry sector will be excluded from charges relating to dam safety inspection). 4. The unit charge for all WRM activities will then be applied to each user's registered volumes to arrive at a WRM charge per user.

Although explicit regulations for Tier 2 bulk water tariffs do not exist, the DWA guideline document is thorough and adequate with regards to full supply cost. However, it is not clear that sustainable use instruments are applied to Tier 2. Thus, guidelines may be required for the setting of bulk water tariffs for non-water board entities such as some Water User Associations and District Municipalities.

Although the revenue generated in the water sector value chain is largest at the municipal level, price regulation in Tier 3 is weak because of combination of ineffective tariffs and lack of ring-fencing of water and sanitation services income. A publication by the WRC reports an under-recovery on potable water and sanitation tariffs (Arcus Gibb (Pty) Ltd, 2010). The study reported that net external cost is not included in the calculation of Tier 3 water service tariffs, and that there is under-provision for depreciation. There is uncertainty within South African municipalities as to the underlying economic rationale of the water service tariff (price) structure (Arcus Gibb (Pty) Ltd, 2010).

In summary, a pure tariff EPI has to consider water resource management across the water value chain. The policy environment for this exists. However, it would likely lead to increased water tariffs and would require suitable ring-fenced institutional arrangements to ensure the tariff income is spent on catchment management activities.

5.3 Industrial Wastewater with Deposit Refund

The structuring of wastewater charges is a fairly complex process, dependent on several contextual considerations, which include the following (DWAF, 2007):

- Type of water uses occurring within a given municipality (i.e. the resulting pollutants that enter the system)

- The classification status of the water resources intended to absorb any untreated effluent and their associated ability to absorb it
- Seasonal variation
- Overarching objectives of protecting water resources. This necessitates charge considerations that promote the prevention of water resource degradation rather than remediation of polluted/damaged water resources.

Municipalities generally require two types of wastewater charges:

a) Flat-rate charges – The flat rate charges are charged to domestic and commercial users, whose wastewater is of a generic quality, and which is linked to their average water use.
b) Variable rate charges – The variable charges are charged to industrial users whose wastewater quality may contain extremes of COD, nutrients and other pollutants.

There are many different regulations on wastewater charges in South Africa, but they are synergistic, and in line with international best practice.

A series of fundamental challenges exist for effective industrial wastewater charge implementation.

Firstly, the implementation of wastewater charges which occurs at the municipal level varies drastically between municipalities (Figure 5-2). In other words, the approach to quantifying the charge (in this case the formula compositions and contaminants considered) drastically varies between municipalities. Furthermore, the formulas are based on historic engineering studies of which the larger metros appear to have well-formulated structures, for both industrial effluent tariffs and the domestic sewerage tariffs, but it is not clear whether the same is true for smaller municipalities. The setting of an appropriate wastewater charge is already a complex and lengthy task requiring consideration of operations costs, infrastructure maintenance as well as economic regulation policy imperatives. As a solution to implementing wastewater charges in smaller municipalities, rapid wastewater charge models have been developed using municipal revenue and expenditure. The wastewater charges need to be indicator specific to the specific industries characteristic of the municipality. Therefore, it would be expected that the approach to wastewater charge implementation would vary, however there must be flexibility within all approaches to be able to accommodate for all indicators (e.g. Mbombela focusses on only CODs as an indicator of wastewater charge).

Secondly, a number of obstacles to the implementation of proper ring-fencing of municipal revenue includes the lack of cost accounting and the highly technical nature of wastewater service provision that make intra-departmental communication difficult. The legislative and policy framework governing wastewater management found that although it is prescriptive, the ultimate responsibility falls to the municipality. The roles and responsibilities similarly place charge structure setting at the municipal level. The regulations governing the setting of wastewater charges are also primarily focused at the municipal level.

eThekwini

Item No.	Description	Present tariff excluding vat	Proposed tariff excluding VAT
2	Disposal of Trade Effluent		
a)	Disposal of Trade Effluent to the sewage disposal system excluding direct discharge to a sea outfall-per kilolitre of trade effluent discharges	4.98	5.32
	Additional of charges for high strength sewage determined in Accordance with the formula: $y(\frac{C}{R}\text{-}1)+Z(\frac{B}{S}\text{-}1)$		
	i) Treatment cost "V" All areas excluding the erstwhile local authority area of Hammarsdale	0.53	0.57
	ii) Treatment cost "Z" All areas excluding the erstwhile local authority area of Hammarsdale	0.49	0.52
	iii) Chemical oxygen Demand Value "K"	Factor 360	Factor 360
	iv) Settleable solids value ''S"	Factor 9	Factor 9

Mbombela

T_i = Cx (Q_i/Q_t) x[0.3 + 0.35 Kc + 0.25 Kn +0.1 Kp]

T_i = Charges due by an individual contributor, R

C = Total cost of sewerage management for both treatment and conveyance and must include fixed, semi fixed and variable charges as set by Sembcorp Silulumanzi (R/month)

Q_i = Sewage flow from an individual contributor, m^3 per month (rolling average

Q_t = Total sewage flow to the Works, m^3 per month (rolling average)

Kc =COD_i/COD_t

Kn =TKN_i/TKN_t

Tshwane

$$Tc = Qct(0.6\frac{(COD\ c - COD\ d)}{COD\ d} + 0.25\frac{(Pc - Pd)}{Pd} + 0.15\frac{(Nc - Nd)}{Nd})$$

T_c = Extraordinary treatment cost to consumer

Q_c =Wastewater volume discharged by consumer in Kl

t =Unit treatment cost of wastewater in R/Kl

COD_c =Total COD of wastewater discharged by consumer in milligrams/litre and is inclusive of both the biodegradable and non-biodegradable portions of the COD

COD_d = Total COD of domestic wastewater in milligram/litre

P_c = Ortho-phosphate concentration of wastewater discharged by the consumer in milligrams of phosphorus/litre

P_d = Ortho-phosphate concentration of domestic wastewater in milligram of phosphorus/litre

N_c =Ammonia concentration of wastewater discharged by consumer in milligrams of nitrogen/litre

N_d =Ammonia concentration of domestic wastewater in milli grams of nitrogen /litre

Cape Town

Tc =X+Y (COD_i/COD_w) + Z+ Penalty

Where Tc =Extraordinary treatment cost to consumer per Kl

X =Conveyance cost per Kl

=C C/V A

Conveyance = The transports of effluent or any liquid waste in the bulk or external lower network from the point of discharge to the inlet of the treatment works

C =The operation and maintenance expenditure towards the conveyance of the waste water in Kl per annum

V A =Adjusted volume (adjusted volume means total volume corrected for infiltration) in Kl per annum

V = Variable treatment cost per Kl

=C T/ V A

Variable = These costs are defined as expenditure that does

Treatment Cost vary significantly with volume and COD leading

CT = the operation and maintenance expenditure

Figure 5-2: Wastewater charge calculations for industrial effluent at four South African municipalities.

5.3.1 Proposed Solutions to Limitations to Effective Industrial Wastewater Charges

Because industrial wastewater charges are currently not accurately calculated, the first step for municipalities would be to understand key contaminants being discharged by major industries. This would enable the identification of key water quality indicators. Once the charge is calculated properly, the tariff would likely increase, although whether it will make industries reduce their pollution is questionable.

The aim of discussing industrial wastewater charge here however, is not to analyse how municipalities can generate more revenue but how can industries and municipalities both benefit and result in less concentrated effluent discharge.

This could be achieved by setting standards for all the identified water quality indicators and further increase the tariff. When industries emit below the limit they will be refunded after every financial year and industries that emit above the limit will be heavily penalized and will forfeit their refund. Thus, a deposit refund mechanism could make industries more elastic to their load of effluent discharge.

5.3.2 Feasibility of Industrial Wastewater Charge with Deposit Refund

This EPI theoretically will be able to address issues related to ecological damage, and industries will be aware of the impact they have on the ecological infrastructure. However, with the current economic situation in South Africa, municipalities encourage industrialisation and overcharging companies for their wastewater discharge will be not be an attractive policy to implement.

5.4 Conservation Credit Trading (Zaalklap Wetland Case Study)

The UN Environment has recently embraced the concept of eco-restoration as a flagship initiative. Various business cases for eco-restoration are currently being investigated as set out in the Kubuqi Desert case and the UN-REDD+ reforestation policy instruments initiatives (UNEP, 2015).

In South Africa, wetland offsets are increasingly being applied in response to pressure from the significant growth and development agenda in South Africa.

From an economic perspective, conservation credit trading as a way of delivering offsets or compensation is associated with a range of purported benefits. In practice, these benefits, while being partly observed and partly perhaps anticipated (but hard to prove), have been instrumental in prompting the establishment of ecosystem asset banking systems (i.e. US, Germany, Australia) (Von Hase, 2013).

Wetland valuation models can be developed to develop the business case of conservation credit trading, which takes into consideration the relationship between the ecological asset (balance sheet item) and the delivery of ecosystem services (income state items) (refer to Figure 5.3). The wetland ecosystem services are dependent on the condition of the ecological infrastructure as well as external factors such as rainfall, land use change, etc. If the condition of the ecological infrastructure is modified, there may be a corresponding change in delivery of ecosystem services.

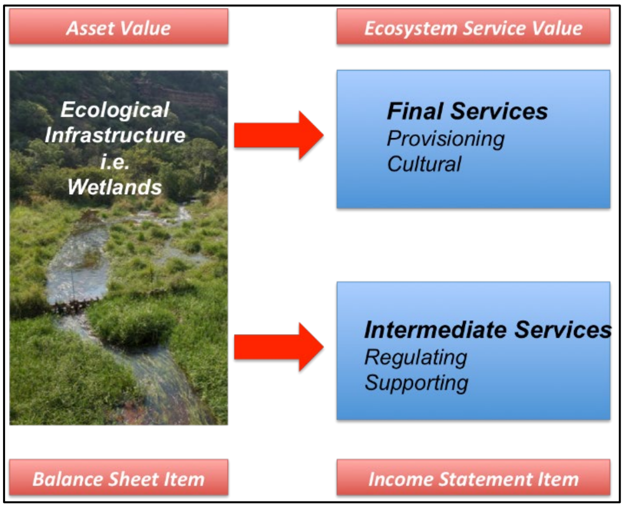

Figure 5-3. The relationship between ecological infrastructure and the delivery of wetland ecosystem services.

The Millennium Ecosystem Assessment (2005) Framework and the TEEB Assessment classify ecosystem services into four categories: supporting, regulating, provisioning and cultural services. The supporting and regulating services produced by wetlands originate from wetlands' role in the biogeochemical cycling and storage of nutrients, organic material and metals and its role as a sink or a source of these compounds depending on the wetland's state and oxygen levels. Sediments are also retained by wetlands. Normal hydrological flux within a wetland and wetland functioning, therefore, has great value in the control of water quality and erosion (Kotze *et al*, 2007).

A comprehensive list of ecosystem services provided by wetlands is given in Table 5-1.

Table 5-1. List of wetland ecosystem services provided by wetlands (TEEB 2013)

Ecosystem Service Category	Ecosystem Service	Ecosystem Structure and Function
Regulating Services	Coastal protection	Attenuates and/or dissipates waves, buffers winds
	Erosion control	Provides sediment stabilisation and soil retention
	Flood protection	Water flow regulation and control
	Water supply	Groundwater recharge/discharge
	Water purification	Provides nutrient and pollution uptake as well as retention, particle deposition
	Carbon sequestration	Generates biogeochemical activity sedimentation, biological productivity
	Maintenance of temperature, precipitation	Climate regulation and stabilisation
Provisioning Services	Raw materials and food	Generates biological productivity and diversity
	Maintains fishing, hunting and foraging activities	Provides suitable reproductive habitat and nursery grounds, sheltered living space
	Grazing opportunities	Provides grazing area for livestock during winter and dry months
Cultural Services	Tourism, recreation, education and research	Provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora
	Cultural, spiritual and religious benefits, bequest values	Provides unique and aesthetic landscape of cultural, historic or spiritual meaning

5.4.1 Conservation Credit Trading

Conservation banking pricing is the value of one hectare of land which considers the value of the expected ecosystem services which will be derived for the duration of the ownership of the land. The following paragraphs discuss the method followed to calculate the value (wetland price).

First, we need to obtain the land price, which is the average cost of 1 hectare of farm land in South Africa. Secondly, we need to obtain the rehabilitation cost of rehabilitating 1 hectare of a wetland. Then we use this cost to calculate the maintenance cost of keeping the wetland in its current state after rehabilitation. This maintenance cost is calculated by taking 10 % of the rehabilitation expenditure per hectare. We lastly need to obtain the ecosystem services value which is offered by a fully rehabilitated wetland. We can then calculate the wetland price by adding the sum of ecosystem service values (which are in perpetuity) to the land price.

5.4.2 Demonstration of Conservation Credit Trading Model – Methodology

Conservation credit trading would provide a valuable mechanism for financially incentivising the protection and maintenance of wetlands in South Africa. Although utilised as an instrument internationally, it is not as clear whether or not a mechanism of this nature would work in the South African context.

Conservation credit trading pricing was thus explored, in a South African context by use of a case model.

Simply put, the price (P) of a wetland conservation credit would equal the sum of the ecosystem service values (into perpetuity), the value of associated land and the cost of wetland rehabilitation (if required per HGM Unit). This value would enable the seller to both recover the capital value of land (whether covered by wetland or not) and acquire additional value through the benefits provided by wetland systems.

An approach that can be taken to applying a conservation credit trading system is demonstrated below.

Conservation credit pricing was determined through the use of the following formulae:

$P = (\sum_{i=1}^{n} PVES_{\infty} + LC)$ *(1+Df)

Where:

P	P: the price of a wetland per hectare
$PVES_{\infty}$:	$PVES_{\infty}$: present value of ecosystem services in perpetuity
LC	LC: land cost per hectare
n	n: number of valued ecosystem values in the wetland
i:	per ecosystem service
Df	demand over-factor (Varies in Scenario 1)

Firstly, the present value of ecosystem services into perpetuity ($PVES_{\infty}$) was quantified. For this demonstration the data for valuation of ecosystem services was mined from a WRC-funded study (Oberholster *et al.*, 2016) done at the Zaalklap wetland on impacts of rehabilitation on ecosystem services. The values for ecosystem services were utilised in this demonstration and are shown in the tables below.

Table 5-2. Value of regulating services per hectare

	Value (Rand/ha)
Water Flow	21 690
Water Purification	6 952
Flood Attenuation	183
Carbon Sequestration	87

Table 5-3.Value of cultural services per hectare

	Value (Rand/ha)
Aesthetic value	206
Recreation	556
Tourism	3 563

Table 5-4. Value of provisioning services per hectare

	Value (Rand/ha)
Harvested products	2 175
Resource poor farmers	9 278
Resource Rent to Agriculture	2 635

It is important to note in this demonstration that the value of the wetland will be affected by the type of HGM Unit. Ecosystem services will be customized in the model according to the wetland type present.

Secondly, the land cost required for conservation credit pricing will simply be the price of undeveloped land per hectare.

Third, the cost of rehabilitation has been sourced from a series of WfW reports (Oberholster et al., 2016).

The rehabilitation cost per type of wetland is shown in Table 5-5, R61 788 is needed to rehabilitate one hectare of a flood plain wetland, R69 945 to rehabilitate one hectare of hillslope wetland and R56 456 is needed for one hectare of a valley bottom wetland. The rehabilitation cost in Table 5-5 is for rehabilitating a wetland from the lowest (25%) level of functionality to the highest (75%) level of functionality. A level of functionality is an indicator of how functional the ecosystem services of a wetland are and these levels are indicated below:

- 0%-25% is a wetland which is in a very poor state with its ecosystem services functioning at less than 25%.
- 25%-50% is a wetland which has its ecosystem services functioning between 25% and 50%. Just below average functionality
- 50%-75% is a wetland which has its ecosystem services functioning between 50% and 75%, just above average functionality.
- 75%-100% is a wetland which is in a very good condition with its ecosystem services functioning at above 75%.

The rehabilitation cost is not included in the pricing of the wetland, but it is an important value adding mechanism to the wetland. This value is captured in the increased ecosystem services.

Table 5-5: Rehabilitation cost of wetland types

Rehabilitation Cost	Rand/ha
Flood Plain	61 788
Hillslope	69 945
Valley Bottom	56 456

An important additional consideration required for the model is the maintenance cost of keeping the wetland in its current state after rehabilitation. This maintenance cost was set at 10% of the rehabilitation expenditure per hectare (WfW approach).

Towards demonstrating the proposed conservation credit trading scheme, three scenarios which have an impact on the scheme will be discussed. These include the following:

- Demand analyses scenario
- ES provided by the wetland scenario
- Wetland state (rehabilitation cost) scenario

5.4.2.1 Scenario 1: Demand Analyses

5.4.2.1.1 Requirements

To ensure the viability of credit banks and to allow these to be competitive in the face of other land use opportunities it is essential to establish a demand for conservation credits. Scenario 1 focusses on the demand for rehabilitated wetlands. An analysis of -50%, 0%, 50% and 100% demand over-factor was utilised. The demand over-factor represents the percentage of demanded area above the supplied area.

For example, a 0% demand over-factor means that the buyers are satisfied with the available area and there is no shortage in the market and no oversupply. A 50% demand over-factor means the buyers still needs 50% more of the current wetland area supply.

The following assumptions were made when populating the model:

- The Land cost is set at R10 000 per hectare
- The rehabilitation cost of R30 894 per hectare for moving a wetland from one functionality level to the next functionality level, here from 25%-50% functionality
- We have the maintenance cost set at R3 707 per annum for one hectare
- Ecosystems services value of R40 833 (provided by selected services provided in Table 5-6)

Table 5-6. The ecosystem services provided by the wetland after rehabilitation

Regulating
Water Flow
Water Purification
Flood Attenuation
Provisioning Services
Harvested products
Resource poor farmers
Cultural Services
Recreation

5.4.2.1.2 Results

The pricing results according to the four demand over factors utilised are given in Table 5-7 and Figure 5-4. The results show the sale of the wetland is profitable from approximately -18% demand over-factor. This means approximately 82% of the rehabilitated wetland area will break even and with increasing demand the profit margin will increase.

In other words, the higher the demand the higher the profitability (assuming supply remains constant) and is indicated by the green triangle in Figure 5-4.

Table 5-7. Demand scenario results

Demand Over-Factor	-50%	0%	50%	100%
Total Cost (Rand)	40 894	40 894	40 894	40 894
Price of Wetland (Rand)	25 417	50 833	76 250	101 667

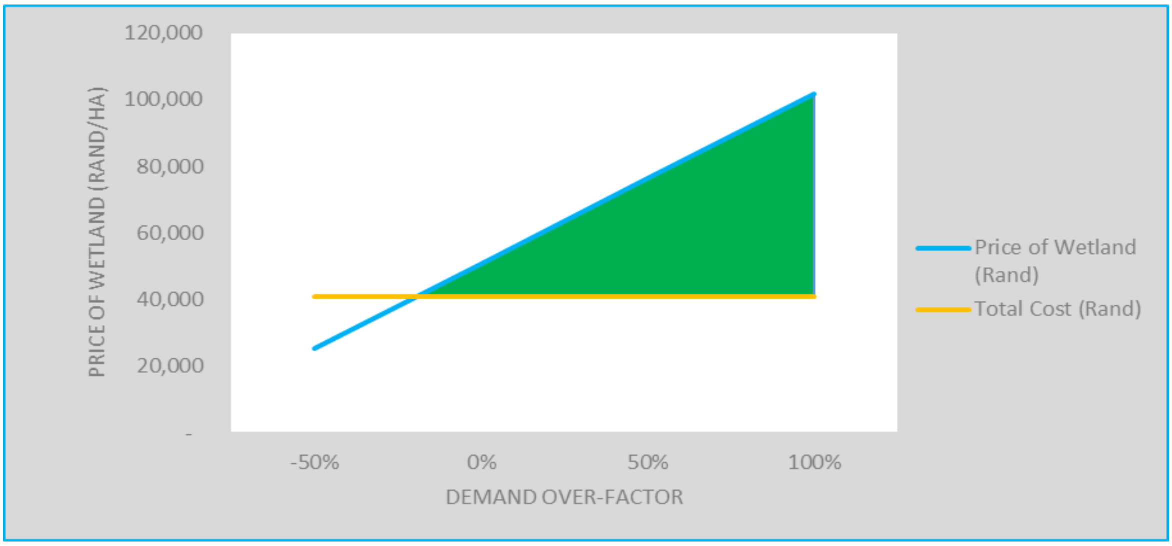

Figure 5-4: Graphical representation of the demand scenario

5.4.2.2 Scenario 2: ES Provided by Wetlands Scenario

Scenario 2 investigates the effects of changes in ecosystem services on prices of wetlands. In other words, it demonstrates the varying price of a wetland conservation credit with regard to varying presence of ecosystem services (For example a wetland in an undeveloped catchment may provide greater/ varied benefits compared to a geographically isolated wetland).

5.4.2.2.1 Requirements

The requirements for this scenario were an understanding of the ecosystem services provided by wetlands as well as the variation of ecosystem services between wetlands and socio-economic classifications of catchments. Towards developing a model to demonstrate this scenario the following assumptions are made:

- The land cost set at R10 000 per hectare
- Maintenance cost of R6 179 per annum for one hectare
- The rehabilitation cost of R30 894 per hectare for moving a wetland from one functionality level to the next functionality level, here from 25%-50% functionality
- Wetlands after rehabilitation provide ecosystems services that are represented in three sets as shown in Table 5-8
- Demand over-factor is set at 0% (supply meets demand equally; the break-even point).

Table 5-8: The set of ecosystem services provided by wetlands in scenario 2

Set 1	Set 2	Set 3
Regulating	**Regulating**	**Regulating**
Water Flow	Water Flow	Water Flow
Water Purification	Water Purification	Water Purification
Flood Attenuation		Flood Attenuation
Provisioning Services	**Provisioning Services**	**Provisioning Services**
Harvested products	Harvested products	Harvested products
Resource poor farmers		Resource poor farmers
		Resource Rent to Agriculture
Cultural Services	**Cultural Services**	**Cultural Services**
Recreation	Tourism	Recreation
		Aesthetic value
		Tourism

5.4.2.2.2 Results

The results show that there are varying prices due to varying presence of ecosystem services in the same unit of wetland area (Table 5-8). In this case it is not necessary to compare the specific set of ecosystem services, but rather it is notable to point out how an ecosystem value contributes to the pricing of conservation credits. Additional and important analyses are shown in Table 5-10 which indicates the individual contribution of an ecosystem service to the price and eventually the cost.

Table 5-9: Development scenario results

	Set 1	Set 2	Set 3
Wetland price including land cost and ES value (Rand)	44 381	50 833	57 238
Rehabilitation Cost (Rand)	30 894	30 894	30 894

Table 5-10: Ecosystem services values (R)

Ecosystem Services (ES)	ES Value (Rand/ha)
Regulating	
Water Flow	21 690
Water Purification	6 952
Flood Attenuation	183
Carbon Sequestration	87
Cultural Services	
Aesthetic value	206
Recreation	556
Tourism	3 563
Provisioning Services	
Harvested products	2 175
Resource poor farmers	9 278
Resource Rent to Agriculture	2 635

With increasing abundance of ecosystem services per ha there will be an increase in the price of conservation credits. Additionally, within ecosystem services, various types provide varying level of benefits and thus possess varying contributions to the value of a wetland. For example, we see that water flow yields higher ecosystem service value than carbon sequestration.

We can therefore conclude that the price of a wetland conservation credit will vary based on the characteristic of the wetland in terms of ecosystem services provided and the nature and classification of beneficiaries.

An important consideration is that ecosystem service value will vary based on its context. Therefore, the assumptions on ecosystem service values made above will not necessarily be appropriate for all wetlands (due to varying socio-economic and functional settings).

5.4.2.3 Scenario 3: Wetland State vs Rehabilitation Cost Scenario

5.4.2.3.1 Requirements

Scenario 3 analysed the effects that rehabilitation costs have on the price of conservation credits. Rehabilitation costs are seen to increase with decreasing health of a wetland and therefore the ability of the wetland to effectively function and provide ecosystem services.

Assumptions required for this scenario include the following:

- Land cost set at R10 000 per hectare
- Maintenance cost of R6 179 per annum for one hectare
- The value of ecosystems services will decrease with decreasing ecological state of a wetland. The assumed value per state is related to ES abundance as presented in Table 5-11
- The rehabilitation cost of R30 894 per hectare for moving a wetland from one functionality level to the next functionality level, here from 25%-50% functionality
- The rehabilitation cost to achieve ecological states are given in Table 5-12
- Demand over-factor is set at 0%
- The rehabilitation cost has a linear relationship with wetland condition

Table 5-11: The ecosystem services provided by each state

State 1 (D- Poor State)	State 2 (C- Average State)	State 3 (B- Good State))
Regulating	**Regulating**	**Regulating**
Water Flow	Water Flow	Water Flow
	Water Purification	Water Purification
		Flood Attenuation
		Carbon Sequestration
Provisioning Services	**Provisioning Services**	**Provisioning Services**
	Resource poor farmers	Harvested products
		Resource poor farmers
		Resource Rent to Agriculture
Cultural Services	**Cultural Services**	**Cultural Services**
	Aesthetic value	Recreation
		Aesthetic value
		Tourism

Table 5-12: Wetland states and the associated rehabilitation cost (for this demonstration we use the cost of floodplain rehabilitation)

Functionality Change	Rehabilitation cost
(75%-75%) (B)	0 (No Need for rehabilitation)
(%50-75%) (C-B)	30 894
(25%-50%) (D-C)	30 894
(25%-75%) (D-B)	61 788

5.4.2.3.2 Results

The results of scenario 3 show that due to the reduced ecological state and rehabilitation requirement of a wetland, the cost of rehabilitation is increased relatively higher than the increase in the conservation credit price of the wetland. This means that a wetland which requires very little rehabilitation to get it to the highest level of quality will be sold at a higher conservation credit price with very little cost involved and thus yielding a higher profit. In this scenario we see that rehabilitating a wetland from the lowest state of functionality has a high rehabilitation cost (R61 788 per hectare) which yields a loss of R14 463 per hectare but a wetland which was already at 50% functionality to a 75% functionality will have a low rehabilitation cost (R30 894 per hectare) and yields a profit of R16 431 per hectare (see Table 5-13 and Figure 5-5). This scenario also shows the discrepancy between the price at which land is sold in the existing market without the inclusion of its ecosystem services' value.

Table 5-13: The results of a wetland state scenario (The price is the sum of all elements)

	State 1 (25%-75%)	State 2 (25%-50%)	State 3 (50%-75%)
Rehabilitation Cost (Rand)	61 788	30 894	30 894
Land Cost	10 000	10 000	10 000
Total Cost	71 788	40 894	40 894
Wetland Price (Rand)	57 325	48 127	57 325
Profit or Loss	-14 463	7 233	16 431

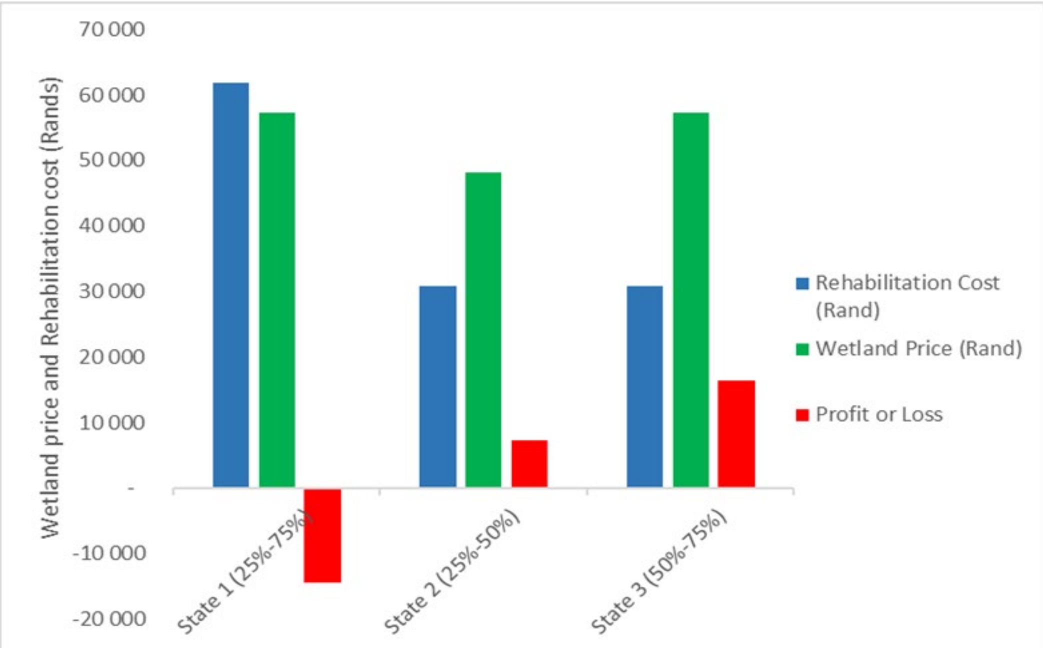

Figure 5-5: The graphical representation of the results of a wetland state scenario

5.4.3 Conservation Credit Trading Model Summary

The preliminary conservation credit trading model investigated scenarios that affect the price of the wetland conservation credits. The first showed that due to the value of wetlands from an ecosystem service perspective, at 0% demand over factor, the sale of wetland credits would (theoretically) be profitable. The second, showed that due to varying spatial and socio-economic settings variations in ecosystem services per wetland would influence the price. Thirdly, the fact that a wetland has re-ceived rehabilitation will influence the price. This is due to the additional factor of cost of rehabilitation and additional functionality received due to the rehabilitation efforts. This analysis provides evidence that a conservation credit trading model, built upon a wetland offset scheme could provide significant incentives for conservation.

5.4.4 Feasibility of Conservation Credit Trading

The conservation credit trading model investigated scenarios that affect the likely prices of wetlands based on their socio-economic context and capacity to provide ecosystem services post rehabilitation. This EPI in the South African landscape has the potential to be implemented successfully as it will be targeted to project developers. However, several conditions exist:

- Ecosystem service benefits need to be clear and measurable
- Users / impactors need to be clearly identified
- Benefits and beneficiaries need to be clearly identified
- An appropriate combination of transaction clearing mechanisms needs to be applied
- Transaction costs need to be controlled
- Creation of high demand
- Model provided evidence that the key factor to successful conservation credit trading is the demand for it. Conservation credit trading would target project developers.

5.5 Waste Discharge Charge System with Tradable Permits (Olifants WMA Case Study)

The purpose of this section is to explore a practical way forward to unpack the WDCS as well as provide complimentary mechanisms that would maximise its influence on users.

The approach includes firstly, assessing the revenue potential of the WDCS. The assessment of the revenue potential of the WDCS was based on the analysis of three primary water polluters:

1. The discharge of wastewater effluents from municipal wastewater treatment works (WWTW)
2. Pollution from agricultural activities
3. The discharge of effluents originating from mining operations.

Secondly, defining an approach to implementing the WDCS through a tradeable permits mechanism. By introducing tradable permits into WDCS, the administration burden for authorities is reduced and a measure of self-regulation is introduced.

The proposed EPI would be that the WDCS would charge upstream polluters and this would incentivise polluters to reduce their effluent discharge loads. Polluters also have the option of mitigating their pollution effects through buying and selling tradable permits. Each water user would have to buy the permit every year and those who cannot emit below the standards can buy from those that emit below the standards. The total tonnes of waste discharge per permit would adhere to maximum allowable resource-directed values (MARDV) (see below).

A limitation of the model is that it considers all pollution loads from downstream users to the Ecological Water Requirement (EWR) sites. Individual waste discharges can therefore not be linked to specific pollution loads. However, by considering the main water users downstream of EWR sites, required load reductions can be linked to specific sectors of water users (in this study either municipalities, agriculture or mining).

The WDCS considers the discharge of wastes with pollutant concentrations that exceed the RQO of identified pollutants to damaging to the system. Central to the WDCS is therefore the setting of:

- Target concentrations (referring to recommended resource-directed value (RRDV) or effluent concentration) as to achieve in-stream resource quality objectives applicable to the resource class
- Maximum allowable resource-directed values (MARDV) in order to achieve the upper limit of the in-stream quality associated with the resource class for each water pollutant.

5.5.1 Methodology

The purpose of the analysis below is to simulate the water pollution charges payable by polluters. This analysis has been designed to include two such charges:

1. An annual permit price (for tradeable permits)
2. A monthly waste discharge charge

Tradable permits are instruments which can be used in combination with the WDCS and do not replace the WDCS. The tradable instrument would be used to control the yearly maximum amount of each pollutant the water user can discharge. The permit can feasibly be sold annually to reduce the administration and the administration related charges of selling it at more frequent or shorter duration. If the water user did not discharge the maximum amount as specified by the permit, they can then sell the permit to another user who requires discharging more than what is allowed by their permit. The following paragraph discusses the method followed to calculate the price of a permit and the loads for each pollutant.

The monthly waste discharge amount is calculated, in principle, by calculating the product of the cleaning cost and the load difference above the REC. To do this, one must first obtain the data on the waste discharge load of the agriculture industry, mining industry and WWTW. The second step is to obtain the cost of removing a tonne of each type of pollutant (SRP, TDS and TIN) from the system. The goal of the WDCS is to recover the mitigation costs and tax component of the WDCS. The clean-up costs in this instance include the cost of labour and capital cost per tonne for removing each pollutant. The price is calculated per ecological water resource (EWR) site. Each EWR site has its own

REC and water users. The principal is thus divided by the total number of the water users per pollutant and billed equally on them. The equal billing of these water users is due to lack of individual's discharges metering which would be used to split the bill according to the individual's discharge.

First, we need to understand the REC of the system and what the WUL allows the holder to discharge. Secondly, we need to determine the number of permits we want to issue. The number of permits issued will be determined by the main objective of the permit, for example, if the objective of the permit is to reduce the waste discharge load in the system, fewer permits can be issued which allow a total discharge lower than that which the system can handle under normal circumstances. The third step is to take the difference between the REC load and the load on the WUL, then divide by the number of issued permits and then multiply by the cost of removing the total pollutants allowed by the permit from the system.

5.5.2 WDCS Revenue Demonstration-Methodology

For this demonstration the data for WDCS revenue potential was mined from Pearce (2014), a WRC study done on WDCS revenue estimation for Olifants WMA.

The first step of calculating WDCS revenue potential is to calculate the load of key water quality indicators at each EWR site per month. To assess the extent of pollutants by the identified waste dischargers, the following water quality indicators were identified:

I. Total inorganic nitrogen (TIN) representing nutrient pollution from partially treated or untreated sewerage discharge and from agricultural practices

II. soluble reactive phosphorous (SRP), representing ortho-phosphates (PO_4^{3-}), used to assess nutrient pollution from sewerage discharge and agricultural activities

III. and sulphates (SO_4^{2-}) to represent the salinity changes due to mining operations (sulphate) due to the impacts of acid mine drainage (AMD)

Each EWR site within the catchment, was assigned a present ecological state (PES) and a recommended ecological category (REC) during the reserve assessment study ("Classification of Significant Water Resources in the Olifants Water Management Area: (WMA 4) – WP 10383", DWA, 2011). The current pollutant load (PES) and the desired load (REC) at each EWR site was calculated by using the upper boundaries of the water quality indicators. The upper boundaries for TIN and SRP, as presented by Scherman Consulting, 2008, were used. The upper boundaries for sulphates were taken from "Olifants River Ecological Water Requirements Assessment, Water Quality Modelling" (DWAF, 2001).

The pollution load within the Olifants WMA was modelled with the load model. The load model calculated the pollution load at each EWR site by using the present ecological state (PES) and the virgin runoff at each site (Equation 1).

Equation 1: Load (kg/month) = PES Concentration (kg/kl) x discharge rate (kl/month)

From the pollution load at each EWR site and by considering the main stem Olifants River and tributaries, the cumulative load was calculated. The load reduction was calculated by decreasing the load at identified EWR sites to meet the REC.

Sections of the river (sections between EWR) as reactors within which load reduction will take place by treatment technologies were considered. The assumption is made that the load within these sections are not reduced or increased by other activities and thus that the load remains constant throughout the section between EWR sites.

Firstly, typical nutrient loads (kg/yr) released can be used to calculate a charge (R/yr) when typical load reduction costs (R/kg) are known. The costing was then based on the principles of the tariff for wastewater.

The cost associated with the removal of sulphates was based on typical treatment costs of reverse osmosis technologies. The calculation of the treatment cost was based on the costs (R/kL) associated with different plant sizes (ML/day). RO treatment plants of 10 ML/day was used which costs 10.72 R/kL. The cost calculations were based on the assumption that 2500mg/L sulphates will be removed during treatment.

These estimates were then utilised to compile an estimate of waste treatment costs for each of water quality indicator based upon the quantity of water that is utilised using equation 2 and 3.

Equation 2 Charge (R/kL) / Concentration Removed (kg/kL) = Charge (R/kg)

Equation 3 Charge (R/kg) x Load to be Removed (kg/yr) = Treatment Cost (R/yr)

The amount to be billed to water users was calculated from the product of the cleaning cost and the load difference above the RQO (Equation 4). This is the value which the water user will be billed for that particular month's effluent discharge. This bill is separated into 2 different parts for a WUL holder operating in the mining sector and for other WUL holders in agriculture and Wastewater Treatment Works (WWTWs).

The WDCS billing price (P) is calculated using the following formula:

Equation 4: $P = (\sum_{i=1}^{n}(AL_i - REC_i) \times C_i)/N$

Where:

P	price to be billed to WUL holder
AL_i:	actual load per WQ indicator
REC_i:	recommended ecological category load per WQ indicator
C_i:	cleaning cost per WQ indicator r
N	number of water users
n	number of WQ indicators measured
i	per WQ indicator (1-SRP, 2-SO_4 and 3-TIN)

The cleaning cost in this instance is the operating cost and includes the cost of labour and capital per tonne for removing each pollutant from water and is shown in Table.

The billing price is calculated per ecological water resource (EWR) site as each EWR site has its own REC and water users. The cost is thus divided and billed equally between the water users. The equal billing of these water users is due to lack of individual wastewater discharge metering or discharge data which would be used to bill the water user according to their waste discharge.

Table 5-14: The cleaning cost per water quality indicator

Water quality Indicators	Rands/tonne
SRP	47 110
SO4	4 290
TIN	9 200

The introduction of WDCS tradeable permits is a mechanism proposed to compliment the WDCS billing process. The approach would be that each water user would need to buy a permit at which point a complying holder of a permit may sell or transfer a permit to a non-complying second party. This provides a waste discharge permitting system that would produce economic drivers to compliance.

The price of the permit could be calculated using the following formula:

Equation 5: $p = (\sum_{i=1}^{n}(REC_i - LL_i) \times C_i)/N$p

Where:

p	permit price. The price at which the permit will be sold at to the water user.
REC_i:	recommended Ecological Category load per WQ indicator
LL_i:	load allowed on the licence per WQ indicator
i:	per WQ indicator (1-SRP, 2-SO_4 and 3-TIN)
C_i:	cleaning cost per WQ indicator for removal in the stream
Np	number of permits issued
n	number of WQ indicators measured

The value (p) obtained here is the price at which the permit will initially be offered to the WUL holders. The number of issued permits (Np) is left as input of the user and this can depend on the permit demand. We will also need the cleaning cost (C_i) of removing each WQ indicator in the river.

The load per permit of each pollutant which would be obtained by the difference between the REC load and the load on the WUL per WQ indicator. This difference will be the total load offered to the water users. This load is divided based on the Np. Each permit will have a MARDV for TIN and SRP in the case of an agriculture-WWTW permit and MARDV for SO_4 in the case of the mining permit.

5.5.3 WDCS Revenue Demonstration Results and Discussion

For the purposes of the demonstration EWR site 1 and EWR site 3 from ("Classification of Significant Water Resources in the Olifants Water Management Area: (WMA 4) – WP 10383", DWA, 2011) were chosen as example data to represent inputs into this demonstration

Demonstration 1: EWR site 1 was represented by the following values and assumptions:

- The SRP load was 0.16 tonnes/month and REC was 0.07 tonnes/month
- The SO_4 load was 679 tonnes/month and REC was 448 tonnes/month
- The TIN load was 5.13 tonnes/month and REC was 2.56 tonnes/month
- The number of WUL issued was 3 in the mining sector and 50 for all agriculture sector and WWTWs

The results indicate a monthly waste discharge charge of R329 830, billed on the mining industry. This is for waste discharges within the MARDV. Should the mining water user require additional allowance to discharge above MARDV, then a mining industry tradable permit to the value of R513 068 (annual) would have to be purchased.

The load was calculated using equation 1 using the present ecological state in each EWR site. Based on the values above, the bill levied on the mining sector is R329 830 per month for each water user and R553 per month for agriculture and municipalities. Based on the EWR site 1 assumptions, the revenue generated amounts to R12 198 713 for one year from the WDCS with 50 combined agriculture and WWTW together and 3 mining water users.

To complement WDCS, the tradable permit system would issue permits amongst water users. The permit would have MARDV for each pollutant. To demonstrate this, a tradable permit system was constructed in this assignment.

The system was based on the same assumptions for EWR site 1 and would issue 10 permits per annum for the agriculture and WWTW sectors and issue 3 mining sector permits. Table 5-15 and Table 5-16 illustrate the agriculture-WWTW permit results and mining permit results, consecutively. Each permit states the MARDV and the price. The price for an agriculture-WWTW permit is R650 per annum and there are ten of these permits on issue, offering the holder to discharge up to 0.00149 tonnes (1.49 Kg) of SRP and 0.05126 tonnes (51.26 Kg) of TIN. The price of a mining permit is R513 068 per annum and there are three of these permits allowing the holder to discharge not more than 30 tonnes (30 000 Kg) per annum.

Table 5-15: The results for an agriculture-WWTW tradable permit

Agriculture and WWTW Permit Price (Rand/annum)	650
SRP Load per permit (tonnes/annum)	0.00149
TIN Load per Permit (tonnes/annum)	0.05126

Table 5-16: The results for a mining tradable permit

Mining Permit Price (Rand/annum)	513 068
SO_4 Load per Mining Permit (tonnes/annum)	30

Demonstration 2: EWR site 3 represented the following values and assumptions:

1. SRP load is 0.07 tonnes/month and REC is 0.02
2. The SO_4 load is 448 tonnes/month
3. The TIN load is 2.56 tonnes/month
4. Number of WUL issued is 2 in the mining sector and 38 for agriculture and WWTWs

The same methodology and approach as per the first demonstration was followed for demonstration 2.

The R411 244 per month billed on the mining water user is for waste discharges within the MARDV. Should the mining water user require additional allowance to discharge above MARDV, a mining tradable permit must be purchased at a price of R293 182 per annum.

The results show that the mining bill would become R411 244 per month for each water user and the bill for the agriculture sector and WWTW would be R442 per month.

Table 5-17 indicates the price and load limits of agriculture-WWTW permit. The price of the agriculture-WWTW permit is R241 per annum and has MARDV of 0.00051 tonnes (0.51Kg) per annum of SRP waste discharge and MARDV of 0.01922 tonnes (19.22Kg) of TIN waste discharge. Table indicates the mining permit results for EWR site 3. The permit price is R293 182 per annum and has

MARDV of SO_4 waste discharge set at 17 tonnes (17 000Kg) per annum. The revenue generated here will amount to R10 071 470 for one year from the WDCS with 38 combined agriculture and WWTW together and 2 mining water users. There is a potential to generate R881 957 from tradable permit with 10 agriculture-WWTW permits and 3 mining permits.

Table 5-17: The EWR site 3 results for agriculture-WWTW permit

Agriculture and WWTW Permit Price (Rand/annum)	241
SRP Load per permit (tonnes/annum)	0.00051
TIN Load per Permit (tonnes/annum)	0.01922

Table 5-18: The EWR site 3 results for the mining permit

Mining Permit Price (Rand/annum)	293 182
SO_4 Load per Mining Permit (tonnes)	17

5.5.4 Feasibility of WDCS with Tradable Permit

The analysis above indicates that the WDCS has the potential to generate large amounts of revenue and would require less administration to conduct its operations compared to the tradable permit system. The downside is that this system would need to evolve beyond equal billing method to the method of billing the water user per their accurately measured discharge. The solution would be better monitoring and measurement of waste discharges.

The tradable permit system on the other hand would need further development on its market and regulation aspects and still needs thorough monitoring of waste discharges. For tradable permits, market development would include the exchange (the administrator handling all the transactions and ownership of permits), the underwriter/clearing house (the handler of unpurchased permits) and the regulator to regulate the market. This would also be technology intensive. The positive side is that at its full development, tradable permits would be the most effective method of controlling pollutants and encouragement of water users to discharge water responsibly.

6 CONCLUSIONS

Currently, South Africa has implemented a predominantly regulatory approach to environmental management to water management.

It is common cause that the South African fiscus is under great pressure, and thus, the required resources are insufficient.

Thus, while we are striving towards a mature regulatory system (as above), we need to be more innovative in finding additional policy instruments that could curb environmental degradation.

Economic policy instruments (EPIs) are such an innovative approach.

In this assignment, we built on existing literature and proposed an environmental policy instrument typology (refer to Table 2-4) that makes a distinction between "fiscal" policy instruments, i.e. relating to Government revenue, and "economic" policy instruments (EPIs), that focus on market creating instruments.

In addition, we made a distinction between EPIs and transaction clearing mechanisms. We defined transaction clearing mechanisms to comprise a set of systems and institutional arrangements, including regulatory policy instruments that are required to make EPIs work. EPIs are therefore specific, policy goal-oriented instruments.

The key output of this assignment is the conceptual framework for EPIs discussed in section 4.1. The framework characterises EPIs by means of six elements. In addition to the transaction clearing mechanisms discussed above, the EPI framework components include:

1. Identification of User / Impactor by Ecosystem Service Category
2. Identification of the key role players, i.e. Regulator / Implementer
3. Identification of specific benefits provided, i.e. the detailed ecosystem services benefits provided, with chains of causality specified according to the MEA and TEEB ecosystem services framework.
4. Identification of beneficiaries – These are two categories of beneficiaries, those that benefit directly from EPI transactions, and could therefore be party to EPI transactions; and those that benefit indirectly through public good benefits.

EPIs have several advantages that make them attractive environmental management tools. These include:

- Reduction in overall cost of reducing environmental damage
- Encouraged use of innovative abatement technologies
- Providing property rights to natural resources to make them valuable
- Self-enforcement by aligning public and private interests

- Increased transparency.

There are however many challenges to implementing EPIs, these include:

- Basic transaction clearing mechanisms and institutional arrangements need to be in place before more sophisticated EPIs can be designed and implemented (refer to section 4.2), and
- EPIs may require increasing prices to reduce consumption or reflect scarcity, which may affect equity and hamper their compliance and effectiveness.

Through applying the EPI conceptual framework, we identify seven possible water EPIs that may be relevant to the South African context. These include:

- A basket of policy relevant (equity, efficiency, sustainability) relevant water use charges
- Green infrastructure management systems
- Eco-restoration
- Waste discharge charges
- Industrial wastewater charges
- Pollution deposit-refund system
- Water pollution permit trading.

Section 5 analysed four of the above EPIs to test elements of the conceptual EPI framework, and to demonstrate various aspects of feasibility assessment required in the design of EPIs.

Our key conclusions are as follows:

(1) If the primary purpose of EPIs is to raise revenue, it would likely fail. Central to achieving a double-dividend outcome, is the concept of Nash Equilibrium, which envisages that role players in a market can each achieve optimal outcomes simultaneously. All role players win, because all get the outcome they desire.
(2) Implementation of EPIs require both regulators and other implementers to be innovative and forward looking. The findings of the 2004 UN Environment indicated that "*lack of understanding of how EIs work; political interests that seek to minimize control costs via regulation; and a preference for keeping the status quo*" hampers effective EPI implementation.
(3) For EPIs to be effectively implemented, a range of suitable transaction clearing mechanisms, and institutional arrangements need to be in place. It is of little use to design sophisticated market-based systems if existing regulatory instruments and their institutional arrangements are not effective (refer to section 4.2).
(4) Effective EPIs require public awareness to be high and uncertainty low. These factors, when combined with a mistrust of regulators and/or firms responsible for environmental degradation, would lead to weak or obstructive participation among stakeholders.

7 RECOMMENDATIONS FOR FURTHER WORK

The domain of EPIs, as defined in this report, allows for highly innovative conceptualisation of specific EPIs to address specific policy objectives. Thus, although the SA landscape of transaction clearing mechanisms and institutional arrangements may not be fully mature, it is possible to design EPIs that build on existing areas of strength.

We therefore propose, for further work, that the WRC identifies EPIs with high likelihood of near term success and conduct in-depth case study-based feasibility studies on these. The case studies would have to:

1. Set clear policy objectives and achieve these
2. Develop, with detailed specifications, the transaction clearing mechanisms required to institutionalise the EPIs.

Perform feasibility testing, using appropriate regulatory and valuation assessment techniques (refer to section 4.2.9).

8 REFERENCES

1. Appleton, A.F. 2002. How New York City Used an Ecosystem Services Strategy Carried out Through an Urban-Rural Partnership to Preserve the Pristine Quality of Its Drinking Water and Save Billions of Dollars and What Lessons It Teaches about Using Ecosystem Services.
2. Arcus GIBB (Pty) Ltd (a). 2010. Literature Review on Pricing and Debt Management for Water Services. WRC Report No 1811/1/10. ISBN 978-1-4312-0030-6.
3. Arcus GIBB (Pty) Ltd (b). 2010. Gap analysis on the status of debt management for water services in municipalities. WRC Report No. 1811/1/11. ISBN 978-1-4312-0029-0.
4. Azrina, M.Z., Yap, C.K., Rahim Ismail, A., Ismail, A., Tan, S.G. 2006. Anthropogenic Impacts on the Distribution and Biodiversity of Benthic Macroinvertebrates and Water Quality of the Langat River, Peninsular Malaysia. Ecotoxicology and Environmental Safety. 64: 337-347.
5. Bakker, A. 2009. Tax and the environment: A world of possibilities, IBFD, Amsterdam.
6. Bank of Israel. 2008. Annual Report 2007. (Hebrew). Bank of Israel, Jerusalem. http://www.boi.org.il/he/NewsAndPublications/RegularPublications/Pages/heb_doch07h.aspx
7. Baumol, W.J. 1972. On taxation and the control of externalities, *The American Economic Review*, 62 (3), pp. 307-322.
8. Bayon, R. 2004. "Will New Regulations Mean Big Business for US Mitigation Bankers?", The Ecosystem Marketplace, www.ecosystemmarketplace.com.
9. Bemelmans-Videc, M.L., Rist, R.C. and Vedung, E. (ed.). 1998. Carrots, Sticks, and Sermons: Policy Instruments and their Evaluation, New Brunswick, New Jersey and London: Transaction Publishers.
10. Bennear, L.S. and Stavins, R.N. 2007. Second-best theory and the use of multiple policy instruments. *Environmental and Resource Economics*, 37, 111-129.
11. Bernstein, J.D. 1997. Water Pollution Control-A Guide to the Use of Water Quality Management Principles- Chapter 6: Economic Instruments.
12. Bertoldi, P. and Huld, T. 2006. Tradable Certificates for Renewable Energy and Energy Savings, *Energy Policy Journal*, 34(2), 212-222.
13. BIS (Department for Business Innovation & Skills). 2011. Advantages and disadvantages of market-based instruments. Crown copyright, U.K. http://webarchive. nationalarchives.gov.uk/þ/http://www.bis.gov.uk/policies/bre/better-regulation-framework/alternatives-to-regulation/choose-the-alternative/market-economic-instruments/ advantages-and-disadvantages-of-market-based-instruments [Accessed 16 August 2012].
14. Bluffstone, R.A. 2003. Environmental Taxes in Developing and Transition Economies. University of Redlands, California, USA.
15. Bovenberg, A.L. and de Mooij, R.A. 1994. Environmental Levies and Distortionary Taxation, *American Economic Review*, 94 (4), pp. 1085-1089.
16. Bovenberg, A.L. and Goulder, L.H. 1996. Optimal Environmental Taxation in the Presence of Other Taxes: General Equilibrium Analysis, *American Economic Review*, 86 (4), pp. 985-1000.

17. Business and Biodiversity Offsets Programme (BBOP). 2009. Biodiversity Offset Design Handbook. BBOP, Washington, D.C. 5 Provincial Guideline on Biodiversity Offsets, Western Cape, 2007.
18. CEC. 2000. Database on environmental taxes in the EU member states. Consultancy prepared by Forum for the Future, July 2001, Brussels.
19. Council of Geoscience. 2010. Mine water management in the Witwatersrand gold fields with special emphasis on acid mine drainage. [pdf]. Available at: http://www.dwaf.gov.za/Documents/AcidReport.pdf> [accessed 14 July 2017].
20. Crafford, J. 2012. Water Benefit Allocation Mechanisms, Water Tariffs, and DWA's Role in a Developmental State: An institutional Economic Analysis. WRC Report. K5-1002.
21. Crafford, J.G., Wilkinson, M.J. Danga, L. 2015. Interim WRC Report: Water Equity Mechanisms for South Africa. Unpublished WRC Report.
22. Dasos Habitat Foundation. 2017. Investing in biological and biodiversity. 1st edition.
23. Department of Water & Sanitation. 2006. Integrated Water Resource Planning Systems Series Sub-series No. WQP1.7.1 Guidelines on Catchment Visioning for the Resource Directed Management of Water Quality Version. April 2006. DWS, Pretoria.
24. Department of Water Affairs. 1994. Water Supply and Sanitation Policy White Paper, Pretoria.
25. Department of Water Affairs. 1997. White Paper on National Water Policy for South Africa. Government Printers: Pretoria, South Africa.
26. Department of Water Affairs. 2007. A pricing strategy for raw water use charge in terms of Section 56(1) of the National Water Act, 1998. Government gazette no: 29697.
27. Department of Water Affairs. 2013. National Water Resource Strategy. Water for an Equitable and Sustainable Future. Government Printers: Pretoria, South Africa.
28. Department of Water and Forestry. 1998. Republic of South Africa. National Water Act. Act No 36 of 1998 Common International Classification of Ecosystem Services. (2012). Retrieved from CICES: http://cices.eu/.
29. Department of Water and Forestry. 1998. The National Water Act, 1998 – A pricing strategy for raw water use charges. Retrieved from https://www.dwaf.gov.za/Documents/Policies/pricing.htm
30. Department of Water and Forestry. 2001. Olifants River Ecological Water Requirements Assessment: Ecological Reserve. Project no. p9104. DWAF Report No: PB-000-00-5299.
31. Department of Water and Forestry. 2003. Strategic Framework on Water Service. Government Printers: Pretoria, South Africa.
32. Department of Water and Forestry. 2004. National Water Resource Strategy First Edition, September 2004.
33. Department of Water and Forestry. 2005. A draft position paper for Water Allocation Reform in South Africa.
34. Department of Water and Forestry. 2007. Strategies for the waste discharge charge system: version 1.9. Pretoria. Department of Water Affairs and Forestry.

35. Department of Water and Sanitation. 2005 a. Summary policy on the Resource Directed Management of Water Quality. Water Resources planning systems series. Sub-series No. WQP 1.5. DWS, Pretoria.
36. Department of Water and Sanitation. 2013. Classification of Significant Water Resources in the Olifants Water Management Area (WMA 4): Management Classes of the Olifants WMA. Report No: RDM/WMA04/00/CON/CLA/0213
37. Department of Water and Sanitation. 2013. National Water Resource Strategy. [Online] Available from https://www.dwa.gov.za/nwrs/
38. Department of Water and Sanitation. 2014. National Water Policy Review – Approved Water Policy Positions. Retrieved from DWA: https://www.dwa.gov.za/Documents/Policies/APPROVED%20POLICY%20POSITION%20 DOCUMENT%20-%2031-01-2014%20TR.pdf.
39. Department of Water and Sanitation. 2015. Revision of the Norms and Standards for Setting Water Services Tariffs in Terms of Section 10 of the Water Services Act, 1997. Gazette no. 39411.
40. Department of Water and Sanitation. 2015. Revision of the pricing strategy for water use charges in terms of Section 56(1) of the National Water Act, 1998.Government gazette no: 39411.
41. Development of a Reconciliation Strategy for the Olifants River Water Supply System. (Final, December 2011) Water Requirements and Water Resources Report. P WMA 04/B50/00/8310/14.
42. Dlamini, X.L. 2015. Resource economic study of the value of the Fountains Valley. City of Tshwane internal study report.
43. Durrheim, R.J., Anderson R.L., Cichowicz, A., Ebrahim-Trollope, R., Hubert, G. Kijiko, A., McGarr, A., Ortlepp, W.D. and van der Merwe, N. 2006. Investigation into the risks to miners, mines, and the public associated with large seismic events in gold mining district. Department of Minerals and Energy, CSIR, Pretoria, 50:1.
44. Edwards-Jones, G., Gotts, D. and McGregor, J.M. 1992. Environmental auditing and its relevance to agriculture. *Farm Management*, Vol 8, No 2, Summer 1992.
45. EEA. 1996. Environmental taxes: Implementation and Environmental Effectiveness. EEA, Copenhagen, August 1996.
46. Food and Agriculture Organization (FAO). 2007. The State of Food and Agriculture. Paying Farmers for Environmental Services. Agriculture Series No. 38, FAO of the United Nations, Rome, Italy.
47. Food and Agriculture Organization of the United Nations (FAO). 2004. Economic valuation of water resources in agriculture – From the sectoral to the functional perspective of nature resource management, Rome, Italy, www.fao.org
48. Fripp, E. 2014. Payments for Ecosystem Services (PES): A practical guide to assessing the feasibility of PES projects. Bogor, Indonesia: CIFOR.
49. Fullerton, D. and Kim, S.R. 2008. Environmental investment and policy with distortionary taxes, and endogenous growth, *Journal of Environmental Economics and Management*, 56 (2), pp. 141-154.
50. Fullerton, D. and Metcalf, G.E. 2001. Environmental controls, scarcity rents, and pre-existing distortions, *Journal of Public Economics*, 80, pp. 249-267.

51. Fullerton, D., Leicester, A., Smith, S. 2008. Environmental Taxes. NBER Working Paper No. 14197 July 2008 JEL No. H23, Q28.
52. Gensler, G. 2009. Remarks of Chairman Gary Gensler, CFTC's Role in Cap-and-Trade, IETA 2009 Fall Symposium.
53. Goldbach, O. D. 2009. Flooding induced seismicity in mines. Paper presented at the 11th SAGA Biannual Technical Meeting, Swaziland.
54. Graham, M., Blignaut, J., de Villiers, L., Mostert, D., Sibande, X., Gebremedhin, S., Harding, W., Rossouw, N., Freese, S., Ferrer, S. and Browne, M. 2012. Development of a generic model to assess the costs associated with eutrophication. Pretoria: Water Research Commission, 2012.
55. Hahn, R. and Stavins, R. 1991. Incentives-Based Environmental Regulations: A new Era from and Old idea? Ecology Law Quarterly, vol 18. 1991. pp1-42. US. Environmental Protection Agency, Environmental investments: The Costs of a clean Environment, EPA 230-12-90-084. December 1990.
56. Helm, D. 2005. Economic instruments and environmental policy. *The Economic and Social Review*, 36(3), 205-228.
57. Hollingworth, B., Koch, P., Chimuti, S. and Malzbender, D. 2011. An investigation into the water infrastructure development financial allocation pathways in municipalities. WRC Report no TT476/10. Water Research Commission, Pretoria.
58. Hood, C.C. and Margetts, H.Z. 2007. The Tools of Government in the Digital Age, Basingstoke: Palgrave.
59. Hosking, S., Hosking, P., Jacoby, K., Sharp, G. and Hosking, J. 2011. Investigating the Mechanism and Processes Used in Setting Water Services Tariffs. Water Research Commission. WRC Report No. 1871/1/11. ISBN 978-1-4312-0121-1.
60. http://www.pmg.org.za/report/20150424-annual-reports-and-financial-statements-rand-water-and-sedibeng-water.
61. International Monetary Fund (IMF). 1998. A Survey by the Staff of the International Monetary Fund.
62. IRMSA, 2017. IRMSA Risk Report, South Africa Risks 2017 Third Edition.
63. Jaeger, W.K. 2002. Carbon Taxation when Climate Affects Productivity, *Land Economics*, 78, pp. 354-367.
64. Jordan, A., Wurzel, R.K.W. and Zito, A.R. 2003. 'New' instruments of environmental governance: Patterns and pathways of change. *Environmental Politics*, 12(1), 1-24.
65. Kachi, A. and Frerk, M. 2013. Carbon Market Oversight Primer, International Carbon Action Partnership, Berlin. https://icapcarbonaction.com/en/?option=com_attach&task=download&id=257
66. Kautto, P. and Melanen, M. 2004. How does industry respond to waste policy instruments Finnish experiences. *Journal of Cleaner Production*, 12, 1-11.
67. Kosoy, N., Corbera, E. 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics, 2010, 69(6), 1228-1236.

68. Kotze, D.C., Marneweck, G.C., Batchelor, A.L., Lindley, D.S. and Collins, N.B. 2007. *WET-EcoServices: A technique for rapidly assessing ecosystem services supplied by wetlands.* WRC Report No TT 339/08, Water Research Commission, Pretoria.
69. Kozmenko, S.; Volkovets, T. 2014. Modeling the environmental taxation: the case of Ukraine. *Environmental Economics*, 5(2), 2014.
70. KPMG. 2012. Cocoa certification: Study on the costs, advantages and disadvantages of cocoa certification commissioned by The International Cocoa Organization (ICCO)
71. Lascoumes and Le Galès, 2007: 5). http://www.ub.edu/sociologia/isdub/pdf/2014/article-1-gales.pdf.
72. LEAF (Linking Environment and Farming). 1994. The LEAF environment audit (1995). LEAF, National Agriculture Centre, Stoneleigh, Warwickshire.
73. Lee, D.R. and Misiolek, W.S. 1986. Substituting Pollution Taxation for General Taxation: Some Implications for Efficiency in Pollution Taxation, *Journal of Environmental Economics and Management*, 13, pp. 338-347.
74. Lenat, D.R., Crawford, K. 1994. Effects of Land Use on Water Quality and Aquatic Biota of Three North Carolina Piedmont Streams. *Hydrobiologia*, 294: 185-199.
75. Lenat, D.R., Crawford, K. 1994. Effects of Land Use on water quality and Aquatic Biota of Three North Carolina Piedmont Streams. Hydrobiologia. 294: 185-199.
76. Lighthart, J.E. 1998. The Macroeconomic effects of environmental taxes: A closer look at the feasibility of “win-win” outcomes. IMF Working paper WP/98/75.
77. Lindfors, H., 2011. “Drinking Water Quality Monitoring and Communication in Rural South Africa”, Thesis, Civil and Environmental Engineering, Chalmers University of Technology, Sweden, Goteborg, Sweden, 2011.
78. Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M. A., Raffaelli, D., Schmid, B., Tilman, D. and Wardle, D.A. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science* 294:806-808.
79. Maila D., Crafford, J., Mathebula, V., Naidoo, N. and Visser, W. 2018. National Water Accounts for South Africa: Systems, methods and Initial Results. WRC Report No K5/2419.
80. Maila D., Mulders, J., Naidoo, N., Crafford, J., Mitchell, S., Harris, M. 2017. An Evidence-Based Approach to Measuring the Costs and Benefits of Changes in Aquatic Ecosystem Services. WRC Report No: K5/2272.
81. Maldonado, V.G.M. 2010. Land Use Influence on Benthic Macroinvertebrate Communities of streams in Nyangores and Amala Tributaries of Mara River, Kenya. MSc Thesis, UNESCO-IHE Institute for Water Research, Delft, The Netherlands.
82. Maldonado, V.G.M. 2010. Land use influence on Benthic macroinvertebrate Communities of streams in Nyangores and Amala Tributaries of Mara River, Kenya. MSc Thesis, UNESCO-IHE Institute for Water Research, Delft, Netherlands.
83. MEA. 2005. Ecosystems and human well-being: current state and trends, 1. Edited by R. Hassan, R. Scholes and N. Ash. World Resources Institute. Island Press, Washington.

84. MEA. 2007. Overview of the Millennium Ecosystem Assessment. http://www.millenniumassessment.org/en/About.aspx 16 May 2007.
85. Millennium Ecosystem Assessment. 2003. Ecosystems and Human Well Being: A Framework for Assessment Millennium Ecosystem Assessment. Island Press, Washington.
86. Mitchell, S.A. and Crafford, J.G. 2015. Independent Review of the Hartbeespoort Remediation Programme. WRC unpublished report.
87. MoEP (Israel Ministry of Environmental Protection). 1997. Professional public committee report to determine the proper procedures for protecting the environment (Hebrew). http://old.sviva.gov.il/Enviroment/Static/Binaries/Articals/vada_report1_0.pdf.
88. MoEP. 2008. Environmental policy and institutional framework of Israel. Ministry of Environmental Protection, Jerusalem.
89. Morgan, B. 2006. Turning-Off the Tap: Urban Water Service Delivery and the Social Construction of Global Administrative Law. The European Journal of International Law Vol. 17 no.1
90. National Treasury. 2016a. Financial Distribution for Government Businesses: Policy and Guideline paper. Treasury Ref: TPP16-04
91. National treasury. 2016b. Capital Structure for Government Businesses: Policy and Guidelines paper. Treasury Ref: TPP16-03
92. Nicolaisen, J., Dean, A., and Hoeller, P. 1991. Economics and the environment: A survey of issues and policy options. OECD Economic Studies No. 16. Spring 1991.
93. NWC. 2011. Water markets in Australia: A short history. Canberra. Government of Australia.
94. NWC. 2012. Impacts of Water trading the southern Murray-Darling Basin between 2006-07 and 2010-11. Canberra. Government of Australia.
95. Oates, W.E. 1993. Pollution Charges as a Source of Public Revenues, in Giersch H. (ed.), *Economic Progress and Environmental Concerns*, Berlin: Springer-Verlag.
96. Oberholster, P.J., De Klerk, A.R., Chamier, J., Cho, M., Crafford, J., de Klerk, L.P., Dini, J.A., Harris, K., Holness, S.D., le Roux, W., Schaefer, L., Truter, J.C. and van Deventer, H. 2016. WRC report No. 2230/2/16.
97. O'Connor, D. 1994. The use of economic instruments in environmental management: The East Asian experience. In: Applying Economic Instruments to Environmental Policies in OECD and Dynamic Non-Member Economies. Organisation for Economic Co-operation and Development, Paris.
98. OECD. 1994. Economic instruments in environmental policy: Lessons from the OECD experience and their relevance to developing economies. OECD Development Centre, Working Paper No. 92 OCDE/GD(93)193.
99. OECD. 2003. The use of economic instruments for pollution control and natural resource management in EECCA. CCNM/ENV/EAP (2003)5. Paris, OECD.
100. Pagiola, S., Zhang, W. and Colom, A. 2008. Assessing the Potential for Payments for Watershed Services to Reduce Poverty in Highland Guatemala. Latin America and Caribbean Sustainable Development Department. World Bank. Washington DC, USA.

101. Palevitch, Y, O Bergerson, Y Arnon and U Mingelgrin. 1998. Economic tools for the protection of the environment. Israel Ministry of Environmental Protection, Chief Scientist Office, Jerusalem.
102. Parry, Ian W. H. & Williams, Robertson III & Goulder, Lawrence H.1999. "Markets" Journal, Elsevier, vol. 37(1), pages 52-84, January.
103. PCE (Parliamentary Commissioner for the Environment) 2006. Changing Behaviour: Economic Instruments in the Management of Waste. Wellington: PCE.
104. Pearce, D., Crafford, J., Burger, A., Pedzisa, T. and Mitchell, S. 2014. An analysis of water pricing instruments governed by the DWA water pricing strategy, and its potential for generating revenue for CMAs. WRC Project K5/2078 Draft Final Report.
105. Perrings, C. 2006. Ecological Economics after the Millennium Assessment. International Journal of Ecological Economics & Statistics (IJEES). Fall 2006, Vol. 6, No. F06; *Int. J. Ecol. Econ. Stat.*; 8-22. ISSN 0973-1385; Copyright © 2006 IJEES, CESER.
106. Perrot-Maître, D. 2006. The Vittel payments for ecosystem services: a "perfect" PES case. International Institute for Environment and Development. Department International Development.
107. Pigou A.C. 1920. *The Economics of Welfare.* Macmillan, London.
108. Pindyck, R.S. 2006. Uncertainty in Environmental Economics. http://18.7.29.232/bitstream/handle/1721.1/45063/2006-017.pdf?sequence=1
109. Qu, X., Wu, N., Tang, T., Cai, Q., Park, Y. 2010. Effects of Heavy Metal on Benthic Macroinvertebrate communities in High Mountain streams. International Journal of Limnology. 46: 291-302.
110. Robertson, M.M. 2004. The neoliberalization of ecosystem services: wetland mitigation banking and problems in environmental governance. Geoforum, Volume 35, 3: 361-373.
111. Rogers, P., Bhatia, R. and Huber, A. 1998. Water as a Social and Economic Good: How to Put the Principle into Practice. Global Water Partnership Technical Advisory Committee. Global Water Partnership/Swedish International. Development Cooperation Agency. S105-25 Stockholm, Sweden.
112. Romans, J.T. 1996. "Moral Suasion as an Instrument of Economic Policy". *The American Economic Review*. 56 (5): 1220-1226.
113. SANBI. 2008. Assessing the Appropriateness of Wetland Mitigation Banking for Securing Aquatic Biodiversity in the Grassland Biome of South Africa. WRC project No KV 200/08.
114. South African National Biodiversity Institute and the Department of Water and Sanitation. 2016. Wetland offsets: A best practice guideline for South Africa. WRC Report TT 660/16
115. Sandmo, Agnar. 1975. Optimal Taxation in the Presence of Externalities. Swedish Journal of Economics. 77. 10.2307/3439329.
116. Seroa Da Motta, R., Alban, T., Saade Hazin, L., Feres, J.G., Nauges, X.C. and Saade Hazin, A. (eds.). 2004. Economic Instruments for Water Management: The Cases of France, Mexico, and Brazil. Cheltenham U.K.: Edward Elgar Publishing.
117. Simonit, S., & Perrings, C. 2011. Sustainability and the value of the 'regulating' services: Wetlands and water quality in Lake Victoria. *Ecological Economics*, *70*(6), 1189-1199. DOI: 10.1016/j.ecolecon.2011.01.017.

118. Smith, A. 1776. An Inquiry into the Nature and Causes of the Wealth of Nations. Edinburgh. Available at www.adamsmith.org/smith/won-index.htm.
119. Sorina-Geanina, M. and Nicoleta-Cornelia, B. 2011. Relevance of economic instruments used in sustainable development process. Annals of the University of Petrosani Economics, 11(3), 155-166.
120. SOUTH AFRICA. 2013. Division of Revenue Bill. Government Printers: Pretoria.
121. SOUTH AFRICA. 1994. White Paper on the Reconstruction and Development Programme (RDP) of 1994. NOTICE NO. 1954 OF 1994. Government Printers: Pretoria, South Africa.
122. SOUTH AFRICA. 1996. The Constitution of the Republic of South Africa Act 108 of 1996. Pretoria: Government Printer. www.info.gov.za/documents/constitution/1996/a108-96.pdf
123. SOUTH AFRICA. 1997. National Water Services Act (Act No. 108 of 1997). Government Printers: Pretoria, South Africa.
124. Stavins, R.N. 1997. Economic incentives for environmental regulation. The New Palgrave Dictionary of Economics and the Law.
125. Stavins, R.N. 2003. Experience with market-based environmental policy instruments. Handbook Environ Econ. 1:359-360, 400, 414.
126. Sterner, T. 2003. Instruments for environmental policy, SIDA.
127. Sunstein, C.R. 2006. Laws of fear: Beyond the precautionary principle. Cambridge University press, March 2005. Available at SSRN: http://ssrn.com/abstract=721562.
128. TEEB. 2010. The Economic of Ecosystem and Biodiversity: Ecological and Economic Foundation. Edited by Pushpam Kumar. Earthscan, London and Washington
129. TEEB. 2013. The Economics of Ecosystems and Biodiversity for Water and Wetlands. IEEP, London and Brussels; Ramsar Secretariat, Gland
130. UNEP Compendium. 2004. On human rights and the environment selected international legal materials and cases: advanced version. Centre for International Environmental Law. Van den Doelen 1987: 53-57.
131. UNEP. 2004. The use of economic instruments in environmental policy: opportunities and challenges. UNEP/ETB/2003/9. United Nations Publication. ISBN: 92-807-2391-X
132. UNEP. 2012. The Role and Contribution of Montane Forests and Related Ecosystem Services to the Kenyan Economy.
133. UNEP. 2015. Review of the Kubuqi Ecological Restoration Project: A Desert Green Economy Pilot Initiative. United Nations Environment Programme, Nairobi.
134. Van Nispen, F. 2011. Policy Instruments. New England Journal of Medicine – N ENGL J MED.
135. Von Hase. 2013. Deliverable 1.1: Preliminary review of international experience with conservation banking Version 0.1 Unpublished draft report prepared for eThekwini Municipality and the Critical Ecosystem Partnership Fund prepared by Forest Trends and Eco-Pulse Environmental Consulting Services.
136. Winter, J.G., Duthie, H.C. 1998. Effects of Urbanisation on water quality Periphyton and Invertebrate communities in a South Ontario streams. Canadian water resource journal. 23: 245-257.

137. World Bank. 1991. Monetary Policy Instruments for developing countries. Caprio, G. and Honohan, P. (Editors). The International Bank for Reconstruction and Development / THE WORLD BANK 1818 H Street, N.W., Washington, D.C. 20433, U.S.A.
138. World Bank. 1997. Five Years after Rio: Innovations in Environmental Policy, Environmentally Sustainable Development Studies and Monograph. Series No. 18.
139. World Bank. 1998. Market-based instruments for environmental policy making in Latin America and the Caribbean – Lessons from 11 countries. World Bank Discussion Paper N. 381.