Economy-wide implications of water quality management policies: A case of the Olifants river basin, South Africa

by

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Declaration

I, Clement Kweku Kyei declare that the thesis, which I hereby submit for the degree of Doctor of Philosophy in Agricultural Economics at the University of Pretoria, is my own work and has not previously been submitted by me to this or any other tertiary institution.

Name:

Signed:……………………………………………...

Dedication

To my late mother, Comfort Laryea for teaching me the “tripod” core values of faith in God, prayer, and hard work

To my future wife and children

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First, I would like to thank the Almighty God for counting me worthy to be part of His creation and for His favour and mercies through this PhD journey.

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Abstract

The deterioration of water quality threatens the functioning of ecosystems and the sustainability of socioeconomic growth and development more especially for a water-stressed country like South Africa (SA). The Olifants river basin, which is one of the nine water management areas in SA, faces serious water scarcity, with declining surface and groundwater quality due to pollution from mining activities, irrigation agriculture, and industrial waste disposal. This has led to great competition for water among different economic sectors and between upstream and downstream users. As a result, the government has implemented a series of pollution control measures with the view to mitigating pollution and water shortage in the basin. This study developed and used a regional, environmental, computable general equilibrium (CGE) model to assess the environmental, economic, and social impacts of protecting the basin's water resources. To calibrate the model, the study also constructed an environmental social accounting matrix (ESAM) using the framework of an environmentally extended SAM.

Results of the environmental and economic impacts of taxing water pollution suggest that internalising the negative externality of water pollution in the Olifants river basin will effectively reduce pollution discharge (i.e., achieve its environmental goals). This, however, comes at some costs to the regional economy of the basin. The economic burden of the tax happens to be fairly small though, due to the small relative share of the water pollution supply and abatement costs in total production costs. Furthermore, recycling the tax revenue

through income transfers to households or a subsidy to pollution abatement mitigates the adverse economic impacts.

Results of the distributional impacts of taxing water pollution in the basin indicate that the water pollution tax is progressive (inequity and poverty-reducing) on the income side as the poorest and vulnerable derive lower shares of their income from capital, which bears the biggest burden of the tax. On the expenditure side, however, the tax is regressive (inequity and poverty increasing), due to the higher share of pollution-intensive goods in poor households' expenditure. The net effect of the tax is, however, not pro-poor. Recycling the tax revenue through a subsidy to pollution abatement sectors reduces the adverse effect of the tax on household welfare whereas uniform direct lump-sum transfers to households' income results in a progressive outcome.

This study has demonstrated the importance of using an integrated framework that allows non-linear substitution possibilities and endogenous price determination to account for both the direct and indirect costs of water quality management policies. The findings should, however, be viewed with caution due to some limitations inherent in basic assumptions of this study. Firstly, demand for pollution abatement services by production sectors, is specified in a simple way using exogenously determined clean-up rates and the assumption that unit costs of pollution abatement are fixed. Secondly, this study did not account for the economic benefits from water quality improvements as well as from technological advancements that lead to reduced pollution intensities.

Keywords: Environmental CGE model, Water pollution tax, Water quality, Distributional impacts, Market-based incentives, Olifants River

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ACRONYMS

AMD	Acid Mine Drainage
CAC	Command and Control
CGE	Computable General Equilibrium
CSIR	Council for Scientific and Industrial Research
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DWS	Department of Water and Sanitation
ESAM	Environmentally extended Social Accounting Matrix
GAMS	General Algebraic Modelling System
IFPRI	International Food Policy Research Institute
IO	Input-Output
IP	Integer Program
KNP	Kruger National Park
LP	Linear Programming
MBI	Market Based Instruments
NLP	Nonlinear Program
NWA	National Water Act
NWRS	National Water Resources Strategy
OESAM	Olifants Environmental Social Accounting Matrix
OWMA	Olifants Water Management Area
PE	Partial Equilibrium
RDM	Resource Directed Measures
RQO	Resource Quality Objectives
RRGDP	Real Regional Gross Domestic Product
SA	South Africa
SAM	Social Accounting Matrix
SDC	Source Directed Controls
UNEP	United Nations Environmental Program
WDCS	Waste Discharge Charge System

WEAP	Water Evaluation and Planning
WMA	Water Management Area
WMS	Water Management System
WQM	Water Quality Management
WRC	Water Research Commission
WRCS	Water Resource Classification System

CHAPTER 1: INTRODUCTION

1.1 Background

South Africa (SA) is a water-stressed country with mean annual rainfall (490mm) below the world's average (814mm) (CSIR, 2010). Furthermore, rainfall is variable and erratic with both spatial and temporal variations. The eastern parts of the country receive a lower amount of rainfall compared with the southern parts (Van Rooyen et al., 2010). Groundwater resources are also not enough to augment surface water due to the predominantly hard rock nature of the country's geology. The Department of Water and Sanitation estimates that about 90% of SA's groundwater exist in hard rocks and is highly variable in nature (DWS, 2011a). In light of these limiting factors, SA has introduced and continues to implement major water policy and management reforms with the intention of protecting, managing and developing the country's water resources. The National Water Act (NWA, No. 36 of 1998) which is considered one of the most comprehensive water acts in the world; attempts to redress past water allocation inequalities and promote the equitable and efficient use of the country's water resources. The act recognises the right of all citizens to basic water supply and the need to manage water resources in an effective and sustainable way due to their scarcity.

As the custodian of the country's limited water resources, the Department of Water and Sanitation has the mandate to manage the water resources in a manner that guarantees sufficient water supplies of reasonable amount and quality for all recognised users (DWS, 2008). In the past, the emphasis was on supply-side approaches such as investment in water supply projects and the use of technical expertise to deliver water and related services. However, with increasing cost of water supply, population growth, and rapid urbanisation, the focus has gradually shifted towards water resource conservation through the use of demand management approaches (Lange & Hassan, 2006; NWRS, 2013). The aim is to enhance the efficiency with which water is presently utilized and minimize loss and wastage of water. Moreover, population growth and rapid urbanisation come with human-induced water problems such as water quality degradation which add pressure to the limited amount of water available. These human-induced water quality problems are associated with agricultural activities which use chemical inputs (e.g. fertilizers and pesticides), industries that discharge chemical waste, poorly functioning sewage treatment works that add excessive nutrients to water resources due to untreated or partially treated effluents, and mines that introduce metals to water resources (DEA, 2011). The major processes contributing to water quality deterioration include salinity (measured as total dissolved salts), eutrophication or nutrient load (measured as total nitrogen and total phosphorus), microbial contamination and sedimentation (CSIR, 2010; DEA, 2011; DWS, 2011a).

The National Water Act (NWA, No. 36 of 1998) provides the legal basis for the protection of water resources through two measures: Resource Directed Measures (RDM) and Source Directed Controls (SDC). RDMs are strategies intended to protect the quality, quantity, in-stream biota and riparian habitat of water resources. It reflects the ecological status and overall health of the water resource. RDM consists of the Reserve (quality and quantity of water for basic human needs and ecosystem functioning), management class (a statement of intent of how the resource custodian and other users want their water resources to be like) and Resource Quality Objectives (RQO) (specific goals set by the resource custodian and other water users for the quality of their water resources) (DWS, 2006).

SDCs, on the other hand, outline the limits and constraints imposed on users of a water resource for the purpose of achieving the objectives of sustainability and equity. They are primarily attached to water use licenses and are guided by the RDM. They include measures such as pollution prevention, effluent discharge control, technologies to minimise waste (abatement) and economic incentives such as levies and fees (DWS, 2006). One economic measure to control impact on water resources is the Waste Discharge Charge System (WDCS), which is an important pricing strategy that considers discharging waste or water containing waste into a water resource another form of water use to be charged for according to the NWA. The aim of WDCS is to internalise the costs associated with waste discharges in accordance with the polluter-pays-principle. WDCS is applied to water management areas (WMA) where there is a significant impact on the quality of the water resources due to human activities (DWS, 2004).

The Olifants Water Management Area (OWMA), which is one of the nine WMAs in SA ranks as the third most water-stressed basin as well as the most polluted (DWS, 2011c; Wambui et al., 2016). However, it is of high strategic importance to the national economy as it supports 48% of the total power generating capacity and contains almost half of SA's strategic water source areas (these areas make up 8% of the land area in SA but provide 50% of the water) (DWS, 2011c; UNEP, 2015). The chronic water deficit and deterioration of water quality faced in the basin are due to increasing water demand and pollution activities from mining, irrigation agriculture and industrial waste disposal (DWS, 2011c).

Water demands in the basin differ across economic activity with agriculture consuming the largest share. In the middle Olifants sub-area, extensive irrigation takes place within the vicinity of the Loskop dam and also near the confluence of the Blyde and Olifants Rivers. Also, the rural population in the middle Olifants, lower Olifants and Steelpoort sub-areas depend to a great extent on the river for livestock production, which serves as an important source of livelihood. The eight major power stations in the country (six of which are coal-fired) located in the upper sub-area makes power generation the second largest water user in the OWMA (DWS, 2011c). In addition, extensive coal mining activities (which supply coal for power generation and export) together with platinum and chrome mines have increased water demand in the basin owing to direct water use and the influx of people into the area employed by the new mines (DWS, 2011c). Eco-tourism is also an important activity in the OWMA. There are a number of private game parks and conservancies and the well acclaimed Kruger National Park (KNP) located in the lower Olifants sub-area (DWS, 2011c).

Each of these economic activities contributes to the polluted state of the Olifants River. Irrigation return flows and seepage, which contains salts from fertilisers, other agrochemicals (such as herbicides and weedicides) and effluent from animal husbandry contribute to the contamination of the river (DEA, 2007). Mining activities in the upper sub-area produce mine water, which is high in dissolved solids such as sulphate, calcium, and magnesium. This contributes to low pH and increased salinity and sediment load which affect in-stream biota as well as riparian habitat. Acid mine drainage (AMD) which emanates from defunct coal mines is also a major contributor to the low pH and high concentrations of dissolved salts characterising the Olifants River. In the upper Olifants also, industrial effluent containing various potential pollutants (such as hazardous chemicals and nutrients) has negative impacts on the quality of the river. Furthermore, untreated and partially treated sewage pumped into the Olifants River by poorly functioning sewage treatment plants compounds the pollution situation of the river.

The effects of these pollutants include widespread eutrophication and toxic water quality, which threatens water supply in the OWMA. Also, the numerous farmers along the river that produce citrus and grapefruits for export are at risk of meeting international standards, because of faecal contamination of their irrigation water. The rural population who use untreated water from the river are also at risk of contracting diseases such as cholera and diarrhoea (De Lange et al., 2012). Therefore, the deteriorating water quality has significant social, economic and health repercussions as well as increase water treatment costs.

The government thus intends to enhance enforcement of the pollution control policies to improve the current water quality situation in the basin and protect the river for ecological sustainability and socioeconomic development. However, pollution control policies have unintended direct and indirect cost implications for economic activities through impacting prices, employment, trade, and income distribution (Xie & Saltzman 2000; O'Ryan *et al.,* 2005; Brouwer *et al.,* 2008). For example, a pollution control policy aimed at minimizing a firm's effluent discharge may have a considerable impact on the activities of other sectors through price changes (arising from increased production cost and expenditure on pollution abatement) that influence demand for intermediate inputs and factors of production in the economy. It is, therefore, important to evaluate the potential social and economic impacts of these policies to inform assessments of the cost-effectiveness of alternative measures and interventions to protect water quality.

1.2 **Statement of the Problem and Motivation of the Study**

The declining water quality in the OWMA due to increased surface and groundwater pollution from mining activities, irrigation agriculture, and industrial waste disposal is of great concern to water resource managers in SA (DWS, 2011c). Not only because of the basins strategic importance to the national economy (supports 48% of SA's total power generating capacity) but also because it is one of the chronic water deficit areas in the country (NWRS, 2013). The area is characterized by low and erratic rainfall patterns (averaging 630 mm per annum with coefficient of variation of above 0.25), and high evaporation rates (averaging 1453mm per annum) which adds to the water pressure by reducing effective rainfall, runoff, and groundwater recharge (Masiyandima et al., 2000; McCartney et al., 2004). This has led to great competition for water among different economic sectors in the basin and between upstream and downstream users (De Lange et al., 2012). Therefore, the upward trend in water pollution in the basin if not addressed will threaten socioeconomic development and aquatic ecosystems. Improving water quality will hence reduce the demand for fresh water and pressure on the current scarcity situation (DWS, 2003).

For this reason, the South African government has adopted a series of measures and programs to mitigate pollution and water scarcity. For instance, the NWA stipulates a Water Resources Classification System (WRCS) – a set of guidelines and procedures when applied to a particular basin will aid in achieving a balance between the need to protect water resources and the need to use them for social and economic development (DWS, 2007). Also, as part of the pricing strategy for raw water use charges (a provision by the NWA), the WDCS has been adopted to control point source pollution in WMA's where the water quality is dire. These measures are underpinned by the Constitution (RSA, Act 108 of 1996), which states that reasonable legislative and other measures should be used to prevent pollution and ecological degradation and to secure ecologically sustainable development and use of natural resources (RSA, 1996). There is no doubt that prudent use of the country's natural resources and pollution control policies could bear important benefits such as efficient resource utilisation, improving water quality, and protecting human health. However, these policies also have cost implications as they impact economic activities through prices, employment, income distribution, and trade. In this regard, assessing the economy-wide impacts and effectiveness of water quality management[1] (WQM) policies becomes paramount given that unnecessarily strict regulation could have negative impacts on the economy.

To address the deteriorating water quality in the OWMA sustainably, policymakers will require detailed knowledge and understanding of the effectiveness of WQM policies and the trade-offs (social, economic and environmental) associated with their implementation. This will need an analytical framework that can quantify the magnitude of the various effects of the policies and thus, select the most cost-effective policy. However, previous studies (DWS, 2003; DWS, 2011b) on the basin-wide impacts of WQM policies have failed to capture the structural realities of the regional economy. Models employed assumed an economy that is linear in costs, and with fixed prices that did not account for impacts on pollution abatement activities. These assumptions do not fit real-world economies, which are characterized by price adjustment and nonlinear substitution possibilities (Xie, 1995). The Computable General Equilibrium (CGE) framework overcomes these limitations by allowing for substitution possibilities in supply and demand systems, and endogenous price determination. Previous CGE applications to water policy reforms in SA however, only focussed on quantity allocations (Mukherjee, 1996; Letsaolo et al., 2007; Juana et al., 2008; Van Heerden et al., 2008; Blignaut & Van Heerden, 2009; Gill & Punt, 2010; Hassan & Thurlow, 2011). Studies that analyse quality dimensions of water management in SA using the CGE framework are, to our best of knowledge non-existent. As a result, the economy-wide effects of water quality management policies in SA remain unclear.

[1] We use water quality management and pollution control interchangeably throughout. The same holds true for the following terns 'economy-wide' and 'basin-wide' and 'pollution' and 'emissions'.

This study, therefore, seeks to fill the above gaps in the literature by adapting a CGE model to include information on water pollution and abatement measures to assess the basin-wide impacts and effectiveness of WQM policies in the OWMA. The approach accounts for both the pollution control cost (i.e. water pollution tax and abatement cost incurred by polluters to achieve the policy goal) and indirect costs (i.e. the impact of the policy on other economic activities arising from behavioural changes in economic agents' decisions in response to implementing WQM policies). The intention is to find the most cost-effective way (with the lowest negative consequences on the economy and social welfare) of reducing pollution and inform and guide water quality policy in SA, particularly in the OWMA.

1.3 Contribution of the Study

This study makes two notable contributions. Firstly, we evaluate the economic, environmental, and distributional implications of WQM policies in the Olifants basin by adapting the IFPRI standard static CGE model developed by Lofgren et al. (2002) to include water pollution and abatement measures of production sectors, water pollution tax, and pollution abatement subsidies. We modify a number of equations in the model to reflect pollution-related costs incurred by polluters as well as government revenue from water pollution taxes. Also, we specify equations describing the cost of pollution control and the total amount of pollution abated and emitted in the economy. In this regard, we integrate the cost of pollution control with a top-down model that determines the indirect costs. These model modifications and additions enable us to capture both the direct cost (i.e. water pollution tax and abatement cost) and the indirect cost (in the form of forgone utility and profits by economic agents) of implementing WQM policies.

Secondly, we construct a new database for the basin that distinguishes pollution abatement sectors from conventional production sectors and accounts for pollution abatement which is treated as a special intermediate input in the production process. To provide new insights and enhance empirical analysis of the reactions of economic agents to pollution control, the theory of pollution control needs to be combined with data to generate results. Therefore, our new database makes it possible to quantitatively analyse the economy-wide impacts of WQM policies in the Olifants river basin.

1.4 Research Questions

The study intends to address the following questions:

1. What are the key sectors with the highest impacts on wastewater discharge in the OWMA?

2. What are the economy-wide implications of water quality management interventions at the catchment level and how to evaluate the trade-offs between pursuing environmental, economic, and social goals of alternative water pollution control and policy measures?

3. What is the most economically efficient and socially optimal way of reducing the level of pollution in the OWMA?

1.5 **Study Objectives**

The main objective of the study is to assess the environmental, economic, and social impacts of WQM policies intended for ameliorating water pollution and scarcity in the OWMA. Specifically, the study will

1. Develop an appropriate model that can capture the direct and indirect environmental, economic, and social impacts of alternative WQM measures for reducing water pollution at catchment level
2. Construct an environmental SAM database for the OWMA that can capture the linkages between economic activities and water pollution and abatement measures.
3. Use the developed model and database to analyse the effectiveness and the economy-wide implications of water quality management interventions in the OWMA
4. Generate scientific knowledge and policy information to guide decision making on the most economically efficient and socially optimal way of reducing the level of pollution in the OWMA

1.6 **Hypotheses**

The following hypotheses are advanced to be tested to achieve the above-stated objectives:

1. *Social (environmental and economic) benefits from reduced pollution will be higher than the costs of pollution control and indirect economy-wide costs associated with introduced measures of water quality management.* This hypothesis is motivated by the two schools of thought in environmental economics with regards to the economic impact of pollution control policies. The first group argues that resource management policies which seek to protect the environment are most likely to harm the economy and reduce employment opportunities. Proponents of environmental regulation on the other hand challenge this view. They highlight the financial benefits of eco-efficiency and the efficient use of scarce resources. We thus test the above hypothesis to add empirical content to this policy debate in the context of the OWMA.

2. *Net social gains from water quality management measures will vary significantly depending on characteristics of the environmental problem at hand as well as the economic and social context of the OWMA.* Inherent trade-offs in environmental policy regulations suggest that no one instrument or government intervention will be suitable for all environmental problems. To the extent that the choice of policy can be influenced by the type of environmental problem as well as political, economic and social factors, net social gains from pollution control will vary accordingly.

3. *The least cost policy for reducing pollution in the OWMA would entail a mixture of pollution abatement measures and economic restructuring.* In other words, pollution-intensive sectors will find an optimal mix between abatement and substitution of pollution-intensive inputs which will lead to a change in the structure of the economy (e.g. a shift from manufacturing to one based on services). This is because the objective of profit maximisation will drive polluters to find least-cost measures in the presence of environmental regulation. As a result, they will search for a combination of inputs and investment in abatement technologies that maximises their profit in a new policy environment.

1.7 Approach and Methods of the Study

This study employs an approach that integrates both economic and environmental activities to assess the environmental, economic, and social impacts of water quality management policies in the Olifants river basin. The approach integrates water pollution related information such as pollutants, pollution taxes, and abatement activities of polluting sectors into a standard static neoclassical CGE model. The CGE model covers the indirect economic costs of the water quality management policies while the environmental component describes the direct environmental costs. To calibrate the model, an environmental social accounting matrix (SAM) was constructed using the framework of an environmentally extended SAM.

1.8 Organisation of the Thesis

The rest of the thesis is organised as follows: chapter two provides a review of relevant literature on analytical approaches and empirical methods employed to study the social and economic implications of environmental policies and related studies. Chapter three develops the analytical model and empirical methods of the study. In chapters four and five, the Olifants environmental CGE model is used to respectively, evaluate the economic and distributional impacts of taxing water pollution in the basin. These chapters present and discuss the associated results and also analyse the trade-offs associated with alternative revenue recycling schemes. Chapter six provides a summary of the thesis, conclusions, and implications for research and policy.

CHAPTER 2: REVIEW OF RELEVANT LITERATURE

2.1 Introduction

Pollution has a long history in economic theory. The problem is considered as the consequence of the absence of a price on emissions, which result in higher volumes than socially optimal (market failure). To address excessive pollution, policymakers employ either Command-and-Control (CAC) regulation or Market Based Instruments (MBI) such as pollution charges and tradable permits. The CAC approach regulates polluters' activity by imposing uniform standards regardless of their efficiency to reduce emissions. MBI on the other hand, use prices and other economic instruments to provide an incentive for polluters to reduce their level of emissions. While CAC allows comparatively little flexibility in attaining policy goals, MBI gives polluters the latitude to choose how much they desire to reduce emissions by comparing the costs of available abatement options with the price imposed on pollution. Thus, polluters undertake pollution control that is both in their interest and collectively meet the environmental policy goals[2]. Implementation of these measures leads to both direct and indirect costs to firms, households, and government. As a result, much effort has been geared towards assessing the implications and effectiveness of these costs and measures. Broadly, these costs are estimated using either partial equilibrium models or economy-wide modelling approaches.

This chapter reviews the relevant literature on the different approaches employed to study the economic implications and effectiveness of pollution control policies. Section 2.2 reviews the literature on partial equilibrium models briefly outlining the underlying theory, its strength and limitations. In section 2.3, economy-wide modelling approaches which consist of input-output (IO), social accounting matrix (SAM), and computable general equilibrium (CGE) models are discussed. Section 2.4 summarises contents of the chapter.

2.2 Partial Equilibrium Models

The Partial Equilibrium (PE) approach is one method used to estimate the economic implications of environmental policies. PE models are simplified representations of real-world phenomenon used to simulate the effects of alternative policies and shocks on a particular sector or market (i.e. part of the economic system represented by one or few products and/or factor markets). The concept was first introduced by Samuelson (1952) and

[2] For more information on the theory underlying the optimal level of pollution/externality and the reasons why MBI's outperform CAC, see chapters 4, 5, and 6 of Pearce and Turner (1990).

later developed by Takayama and Judge (1974) using their spatial and temporal equilibrium framework (Productivity Commission, 2010). PE models are based on neoclassical theory, which assumes all markets clear and economic agents aim to maximise their payoffs by equating marginal benefits and marginal costs (Bouman et al., 2000). Therefore, PE models portray behavioural relations that underpin outcomes in the considered market by tracking the effects of policy changes (such as pricing, investment, and pollution control) on production and consumption decisions. Due to their relatively small size, PE models have the advantage of modelling in detail the supply and demand of the commodity under analysis. This enables the analyst to find the best functional form and to improve the estimation of functional parameters (Dellink, 2005). However, this advantage is also the major drawback of PE models. They assume that the prices of other commodities or factors are fixed and that changes in the considered market have no effect on the rest of the economy (Dudu & Chumi, 2008).

Mathematical programming is one PE technique commonly employed to find the least-cost way of protecting environmental quality. The method optimizes an objective function (such as cost or social welfare) subject to a set of constraints including production, consumption and environmental quality standards. The objective function, constraints, decision variables, and bounds on variables are together known as the basic ingredients of a mathematical program. Based on these basic ingredients, mathematical programs are classified as either linear program (LP), integer program (IP) or nonlinear program (NLP).

Most mathematical programming models applied to water quality control relates to stream pollution. These models employ the Streeter-Phelps (1958) differential equations to describe the removal of biochemical oxygen demand (BOD) or the increased concentration of dissolved oxygen (DO). Although the flow equations vary across models, the mathematical programming structure remains the same. Saremi et al. (2010) used a linear programming model to estimate the minimum treatment cost required to reduce BOD load concentration at different monitoring sites along the Haraz River in Iran. They employed the Streeter-Phelps equation to predict the steady-state BOD load concentration at each river reach. The authors found that the minimum treatment cost differs across pollution sources due to the non-uniform removal rate of BOD at these sources. In a similar study, Mahlathi et al. (2016) found that in-stream DO standards can be met at minimum cost for the Olifants River catchment when treatment levels are optimized simultaneously at each treatment plant. Their study used a mixed integer programming model together with a Streeter-Phelps model which was used to forecast the in-stream DO profiles in response to varying wastewater discharge regimes. Other studies (Mo et al., 2015; Li et al., 2015) have used more complex optimization algorithms such as the interval two-stage stochastic integer programming and modified fuzzy credibility constrained programming approaches. These optimization approaches were employed to study the management of urban and agricultural water resources in China.

Another area where the mathematical programming approach has been used extensively is air quality control. Atkinson and Lewis (1974) employed a linear programming model to compare three alternative approaches – ambient least-cost (ALC), emission least-cost (ELC) and State Implementation Plan (SIP) – to achieve ambient air quality standards at minimum cost for the St. Louis air quality control region. The ALC approach uses individual source marginal control costs and emission dispersion characteristics to compute the allowable source emissions at least-cost while ELC assumes that a unit emission will have the same impact on ambient air quality regardless of source. The SIP approach, on the other hand, ignores both the individual marginal control costs and emission dispersion characteristics. The study found that an air pollution control strategy that accounts for both source specific marginal control cost and emission reduction leads to substantially lower pollution control costs. As a result, the ALC approach produces the least-cost solution to achieve ambient air quality standards. This finding was corroborated by Seskins et al. (1983) in a study that examined the costs of meeting nitrogen dioxide (NO_2) emission standards under alternative emission control strategies for Chicago using an integer programming model. The authors also showed that the minimum cost strategy accounts for both the sources' incremental costs of air pollution control and incremental contributions to ambient pollution concentration. In addition, their study found that market-based strategies (such as emissions charge plans and marketable permits) are more cost-effective because they provide incentives to firms to develop and apply new emission control technologies. In a related study, Krupnick (1986) employed the same approach and software used by Seskins et al. (1983) to estimate the cost of meeting NO_2 standards in Baltimore. However, in the Krupnick study, an additional control strategy and alternative ambient air standards were analysed. The author found that the least-cost policy charges source specific effluent fee and combines regulatory and market-based policy features. In another study, Perl and Dunbar (1982) used a linear programming model to estimate the cost of alternative sulphur dioxide (SO_2) regulations which accomplish the same emission reductions at minimum cost in the United States. The study also compared the cost of regulating SO_2 emissions with the benefits of SO_2 controls. They found that instituted air pollution regulations (mostly command and control type regulations under the Clean Air Act) result in much higher costs, and possibly dirtier air, than required. The higher cost of the Clean Air Act was due to inefficiencies of source-specific emissions limits and technological constraints imposed on new sources. With regards to the cost benefit analysis of SO_2 emission regulation, the study found that economic policy approaches produce higher net social benefits than command and control policies.

The mathematical programming approach has also been used to study the optimal allocation of water resources among competing users including agriculture and mining. Lévite et al. (2003) used the Water Evaluation and Planning (WEAP) model which is based on an iterative linear programming algorithm to analyse the impact of various water allocation scenarios in the Steelpoort sub-basin of the Olifants River. The study revealed that some water users in the basin are not able to meet their requirement and during certain years, the mandatory ecological reserve is also not met. The study concluded that water demand management procedures hold the potential for remedying the water scarcity situation in the Olifants river basin. In a related study, McCarthy et al. (2007) also used the WEAP model to simulate water demand among alternative users over a 70-year period of varying rainfall and river flow. The authors estimated the annual economic cost of failing to supply water to be between USD6 and USD50 million. They concluded that this cost is likely to increase significantly if water demand increases in the Olifants river basin are not checked.

The preceding empirical findings provide support to the received theory that market-based strategies of pollution control are efficient and effective compared with command and control approaches. In addition, these studies have shown that a differentiated approach that accounts for source-specific marginal control cost and emission reduction achieves environmental control standards at minimum cost. Notwithstanding, the cost-effectiveness of pollution control strategies is also influenced by the type of pollutant and the meteorological conditions. However, the method adopted by these studies mostly relies on the firm's response to environmental regulation using specific technology options given by engineering experts. This becomes problematic when applied at a broad scale because technologies differ across firms and locations. Again, although the studies provide useful insights and cost estimates associated with environmental quality control, they fail to account for spillover effects on other commodities and factor markets. This becomes vital when environmental policy reforms lead to non-marginal changes in the economy.

2.3 Economy-wide Modelling Approaches

The economy-wide or multisector modelling approach overcomes the limitation of partial equilibrium models, by accounting for the interdependencies that exist between the different markets and agents in an economy. The essence is to show that changes in government policy or random shocks have far-reaching impacts on the entire economy and are not confined to only one economic sector. The approach began with the pioneering work of Wassily Leontief on multisector analysis which led to the development of his input-output framework in the early 1950s. Economy-wide modelling approaches consist of input-output (IO), social accounting matrix (SAM), and computable general equilibrium (CGE) models.

2.3.1 Input-Output Models

Input-output (IO) models are a convenient framework for representing the interdependence that exists between the different sectors (especially, the productive sectors) of an economy. The framework was pioneered by Wassily Leontief (1953) and is a powerful tool for national and regional economic planning where they are employed to explain and predict the behaviour of the economy. IO models assume a linear structure between sectors with the output of each sector either satisfying final demand and/or used as an intermediate input in the production of other goods. The data for an IO model is captured in a table with rows representing the supply of goods and services and columns tracing the sources and magnitude of inputs required to produce these goods and services. The data represent an economy at a specific point in time (usually a year) and generally in monetary terms. The number of industries considered in an IO model depends on the objective of the study and data availability however, it may vary from only a few to thousands. IO tables are used to generate multipliers needed to estimate the economy-wide impact resulting from a change in final demand sectors (such as an increase in export demand or government spending). Standard IO models can be extended to include environmental data which makes it possible to quantify environmental loads associated with production activities in an economy (Wang et al., 2009). These models have been used to analyse resource usage and environmental pressures in different economies across the globe.

IO models for studying the interdependence between the economy and the environment include works by Daly (1968), Isard et al. (1968), Ayres and Kneese (1969) and Leontief (1970). These models treat the interaction between the economy and environment as totally closed compared with the open manner used by formal policy analysis. The intention is to uncover the interactions between the economy and the environment and assess how policy alternatives modify the two-way flow process (Forssell & Polenske, 1998). However, a major challenge for these early models concerns internal accounting consistency (i.e. how the internal construction of the model influences the consistency of the accounts). Daly's (1968) model integrates economic sectors (production, consumption, and distribution) with environmental sectors (ecological goods). Although his model was useful for descriptive purposes, it could not be used for analytical reasons because he incorporated ecological goods, which have no market prices, with economic goods, which have a market price (i.e. the units were not comparable). The problem of non-comparable units was resolved by Isard et al. (1968) using the commodity-by-industry set of accounts, which allowed industries to produce both their own products and an associated waste product.

In both Daly and Isard et al.'s models, total emissions generated were associated with economic activity, while 'goods' derived from the environment were shown as inputs into the economic sector. Albeit intuitive, this approach presents a problem to the analyst who finds it difficult to assign monetary values to the environmental sector. Again, the traditional industry-by-industry accounts used in Daly's (1968) work results in joint products along each row, leading to aggregation problems. Leontief and Ford (1972) provided an insightful solution to the internal consistency problem. They reversed the position of the environmental inputs and outputs such that the various pollutants have a row in the matrix, so pollution produced by each industry is assumed to be a function of output. Additionally, the authors appended to the columns of the IO table, a set of pollution abatement sectors which demand intermediate inputs from the conventional producing sectors. Their transformation helped resolve the joint-product problem in that analysts could technically solve for pollution as a function of each industry's output.

Recent applications of IO models to study river water quality management include works by WRC (2000), Resosudarmo (2003), Sánchez-Chóliz and Duarte (2005), Okadera et al. (2006), and Qin et al. (2014). Resosudarmo (2003) extended an IO table of Indonesia to include water pollution quantity and abatement cost accounts. The objective was to study how water quality policies could help improve river water quality while at the same time maintaining the growth of economic activities. The author developed several indices (such as effective pollution prevention and effective pollution abatement) to analyse the relationship between production activities and pollution quantities as well as pollution abatement costs. The results show that sectors (such as rubber and plastic and other manufacturing) with high parameters' values of effective pollution abatement should be given incentives in the form of pollution abatement subsidies to clean up their pollution. This is because it is comparatively cheap to clean pollutants from these sectors. On the contrary, consumers should be discouraged (by applying relatively higher sale taxes) from increasing their demand for outputs from sectors with high parameters values of effective pollution prevention. For the reason that increasing demand in these sectors leads to relatively large amounts of river water pollution that is expensive to clean up. Sánchez-Chóliz and Duarte (2005) arrived at a similar conclusion in their study of water pollution associated with the production structure of the Spanish economy. Their results show that a socially beneficial policy (i.e. a water quality policy that leads to lower opportunity costs in terms of income, of stricter pollution standards) is one that reduces demand for pollution-intensive goods but at the same time raises demand for non-polluting goods. Additionally, the authors found that technological advancement holds the potential to reduce pollution levels while improving economic outcomes. Also, Qin et al. (2014) found that there is a potential to reduce fresh water demand and river water pollution by restructuring economic sectors in the Haihe river basin in China. Their study applied a hybrid IO model that integrated economic and ecological systems to analyse water consumption and wastewater discharge.

In South Africa, the Water Research Commission funded research to estimate the economic cost of salinity to the economy (WRC, 2000). The study employed a methodology that integrated direct, indirect, and induced costs as well as behavioural changes by agents in the economy. The direct cost for each sector was estimated using a set of linear equations while the indirect cost was estimated using the IO framework. The estimated direct cost from each sector was then introduced as exogenous data in the IO analysis. This approach allowed the calculation of salinity multipliers' which measures the impact of additional production on the cost of combating salinity. In addition, an IO pricing model was used to show how an increase in a sector's direct salinity cost is distributed across the economy. The results show high spin-off effects of increased salinity. As a result, the study concluded that direct cost is a poor reflection of the cost impacts of salinity. Moreover, the direct impact of salinity abatement was found to be relatively small.

The IO framework has also been employed extensively to study water allocation and consumption patterns. Chen (2000) extended the standard IO model by introducing water inputs as production factors to study the balance between supply and demand for water resources in the Shanxi Province of China. The study concluded that it is better for Shanxi

Province to restructure its economy and reduce the production of goods with high water input coefficients such as rice. Velázquez (2006) employed an IO model to analyse the impact of sectoral water consumption on the economy of southern Spain. The model included both economic and environmental indicators, which were used to analyse regional impacts of various water management scenarios. The study revealed that in planning the productive economy of a region, total water consumption (direct and indirect) by sectors is a key parameter to consider. Wang et al. (2009) developed an IO model based on the approach adopted by Velázquez (2006). Their results show that the agricultural sector (forestry, farming, fisheries, and animal husbandry) demands higher amounts of water directly compared with the industrial and service sectors. Hassan (2003) applied a quasi-input output model in the Crocodile River catchment in South Africa to estimate the direct and indirect benefits (total economic value) from the agricultural sector with a focus on irrigated crops and cultivated plantations. The results show that policymakers should consider both the direct and indirect benefits as well as multisectoral linkages when evaluating the social worthiness of the different productive uses of water and other economic resources.

The studies reviewed above provide valuable insights into the relationship between economic activity (production, consumption, and exchange) and the environment (consumption of natural resources and pollution). We have seen that due to the interrelations within the economy, any economic activity in the form of a policy change will have a positive or a negative impact on the environment. Thus, to reverse the increasing pressure on natural resources and improve environmental quality, policymakers must enact policies that lead to economic restructuring and adoption of technologies with lower pollution discharge per unit of output. Furthermore, the agricultural and manufacturing sectors have been found to be major users of water and culprits in river water pollution. Notwithstanding, the analytical approach employed by these studies suffer from inherent limitations such as the assumed linearity between inputs and outputs, no supply constraints, and fixed relative prices. Moreover, IO models do not account for the interdependence between the productive sphere of the economy and the household sector. That is, the approach does not allow income distribution and expenditure of households to feedback into the economic system (Matete, 2006). This becomes important when the focus of the study is on households or when policy changes significantly impact households' welfare.

2.3.2 SAM Based Models

SAM-based models extend the IO framework to include not only interindustry transactions but also income generation, distribution, and spending. In this regard, they provide a complete perspective on the circular flow of income in the economy and are a powerful research tool for analysing social and economic policy. The concept of social accounting matrix as an accounting framework was first introduced by Sir Richard Stone in the 1960s. The idea was further developed into a modelling framework in the 1970s by Pyatt, Thorbecke, Round, and others to study issues such as economic growth, income distribution and poverty, particularly in developing countries (Round, 2003). The overriding feature of a SAM is the inclusion of the household sector which can be disaggregated into different groups based on a number of characteristics such as main income source (farm/non-farm), location (urban/rural), and household head (gender and employment status). As a result, SAM-based models are advantageous when analysing the income effects associated with a policy change or an exogenous shock on different household groups.

Existing literature on the application of SAM-based models to study environmental quality control and resource management include works by Resosudarmu and Thorbecke (1996), Weale (1997), Xie (2000), Manresa and Sancho (2004), Lenzen and Schaeffer (2004), Morilla et al. (2007), and Cardenete et al. (2012). Resosudarmu and Thorbecke (1996) developed a procedure to modify and extend a SAM to incorporate the link from the economy to the environment, and additionally feedback from the environment to the economy. Their method treats pollutants as by-products of production activities and considers the health costs incurred by the government and individuals resulting from excessive pollution as societal environmental costs related to pollution. Using outdoor air pollution in Indonesia as a case study, the authors analysed the impact of policies designed to improve air quality on the incomes of 10 household groups disaggregated by location (urban and rural) and income level (high and low). They simulated three policies designed to decrease the quantity of air pollutants under two settings: optimistic and pessimistic settings. The optimistic scenario assumes that air quality improvements can be achieved without a reduction in output of pollution-intensive sectors due to the adoption of available technologies, while the pessimistic scenario presumes that implementing air quality regulations will lead to output reduction of these sectors. They found that if technologies available to reduce the quantity of pollutants in the air are at moderately cheaper cost, then air quality control policies will positively impact income distribution. Otherwise, regulations designed to improve air quality will have a significantly negative impact on household incomes and result in unequal income distribution. Weale (1997) found that shifts in exogenous demand have the largest environmental effects in sectors (such as oil extraction and agriculture) with direct impact on the environment. His study extended an Indonesian SAM to include three types of environmental-resource linkages: land degradation, timber harvesting, and crude oil depletion. Cardenete et al. (2012) also found that production activities (especially coal extraction and refined oil) are responsible for the bulk of total CO_2 emissions in the Spanish economy. As a result, they suggested the adoption of energy-saving technologies to reduce CO_2 emissions.

Xie (2000) developed a conceptual framework for extending a SAM to capture the interactions among economic activities, pollution emissions, and pollution control activities. His framework distinguishes conventional production activities from pollution control activities, accounts for pollution control subsidies, environmental investment, and treats pollution cleanups as special intermediate inputs bought by production sectors. Using the framework, he constructed an environmental SAM for China and employed a SAM multiplier and structural path analyses to assess the environmental impacts of Chinese economic policies. The results show that changes in exogenous demand have important impacts on pollution-related activities (such as pollution generation and pollution abatement), and thus an environmentally extended SAM is a useful tool for understanding the environmental effects of economic activities. In a similar study, Lenzen and Schaeffer (2004) constructed an environmentally extended SAM for Brazil and employed different SAM multipliers to assess the trade-offs between economic, social and environmental policies. They found that due to unequal ownership of means of production, the income distribution pattern of production in Brazil has changed, towards extending the income gap between rich and poor. They concluded that though there is scope for enacting policies to bridge the income gap and reduce environmental burden, such policies must promote structural change from resource-intensive to value-adding production. Morilla et al. (2007) also found that there is scope to design environmental policies that improve both the environment and enhances economic growth. Their study integrated economic flows with physical water flow and atmospheric emissions to analyse the efficiency of different economic activity in the Spanish economy.

Even though SAM-based models are an improvement over the IO approach, they are based on restrictive assumptions such as fixed prices and do not incorporate supply constraints or substitution possibilities. As a result, they are appropriate for analysing the impacts of short-run and/or small-scale policy changes. The next section reviews CGE models which allows for substitution possibilities in supply and demand systems, and endogenous price determination.

2.3.3 Computable General Equilibrium Models

CGE models are models with foundations in general equilibrium theory (pioneered by Walras (1874) and later developed by Arrow and Debreu in the late 1940s). The theory of general equilibrium (which depends on the fundamental observation that economic agents and markets in real-world economies are interconnected) provides useful insights into factors and mechanisms that determine relative prices and resource allocation patterns in actual economies (Bergman, 2005). In this manner, CGE models serve as a powerful research tool in the analyses of resource allocation and income distribution issues in market economies.

A CGE model is basically a simplified representation of an economy that incorporates behavioural assumptions (such as utility and profit maximization) about its agents. It is characterised by different production sectors and a final demand sector, which often includes households, government, and exports. The production sector demand factor inputs from households (land, labour, and capital) and produced inputs from other production sectors (intermediate inputs) to produce output. Factors of production are bought from the factor markets and outputs are sold in product markets. It is assumed that flexible relative prices simultaneously clear both the product and factor markets. Households earn rent in return for providing services of their factor endowments and spend their income to purchase commodities as well as paying taxes. The government collect taxes (direct and indirect), buy commodities from the product market and redistributes some of its revenue from taxes to agents in the form of social grants and subsidies. The notion of cross-country dependence is captured in CGE models by specifying trade flows between the modelled country and the rest of the world.

Johansen's (1960) Multisector Growth Model (MSG) is generally accepted as the first empirical implementation of a Walrasian general equilibrium system. Although CGE models started in the 1960s, its extensive application to analyse topics such as tax policy, international trade, natural resources, and the environment began in the 1970s. Currently, there is a wide variety of CGE models employed to analyse various issues both in developed and developing countries. Classification of these models depends on the underlying economic theory, how the behavioural parameters are estimated and the time dimensions included in the model. With regards to economic theory, Robinson (1991) distinguished between neoclassical and structural CGE models. The former is based on neoclassical theory and mostly focused on issues of structural adjustment while the latter is based on political economy theories and concerns the structural characteristics of the economy. Based on the time dimension, CGE models can either be static or dynamic. Static models pay more attention to the interrelations that take place within the economy during a given time period whereas, in dynamic models, investment and savings decisions become important to connect savings and investment in the initial time period with capital formation. In relation to the technique used to determine the behavioural parameters, we can distinguish between CGE models with parameters based on the calibration technique and those with parameters based on econometric estimation. CGE models can likewise be divided into single-country, multi-country, and global models.

In the field of environmental economics, CGE models have been used to analyse the external effects of production and consumption activities as well as management of natural resources. The primary purpose is to quantify and explain the impact of policies aimed at internalizing externalities or improving the management of natural resources. Early environmental CGE models focused on the impact of energy policies on energy supply and demand, mainly in connection with the oil price increases in the 1970s. The model developed by Hudson and Jorgensen (1974) for energy policy analysis in the U.S. is considered the first environmental CGE (Bergman, 2005). It was essentially an energy sector model wherein the rest of the U.S. economy was represented by an exogenously determined rate of growth of energy demand. The model parameters were estimated using the econometric technique. However, the early parts of the 1990s saw a pattern of progress from using environmental CGE models for energy policy analysis to climate change policy analysis (Burniaux et al., 1992; Murthy et al., 1992; Nordhaus, 1994; Hill, 2001). The main reason for this switch was the concern about the external effects of the use of fossil fuels, particularly acid rain and carbon dioxide. Greenhouse gases (GHG) such as carbon dioxide and nitrous oxide were shown to be causing significant changes in the world's atmosphere. As a result, a well-designed environmental CGE model was deemed valuable in elucidating the economy-wide impacts of climate change policies.

In general, environmental CGE models differ in the way they link economic activity and the environment. They can be grouped based on how they model pollution and pollution abatement activities. The first group of environmental CGE models assumes that pollution bears a fixed relation with sectoral output and have their foundations in Leontief's stylized environmental IO model[3]. Dufournaud et al. (1988) developed an environmental CGE model to compare the effects of two alternative ways (an indirect tax on polluting sectors or an income tax on households) of financing pollution-cleaning services. They calibrated their model parameters on Leontief's stylized data and modelled production and consumption activities with a CES production and Cobb-Douglas utility functions, respectively. They assumed that the government pays for the services of pollution removal with revenues raised by taxes and that there is no private demand for pollution cleaning services. In addition, they assumed that the optimal level of pollution is zero (i.e. no pollutants are discharged). Their analysis revealed that CGE models which incorporate substitution and endogenous prices provide more realistic answers to environmental questions than the IO framework.

Robinson (1990) is another study that calibrated his model parameters on Leontief's stylized data to analyse the optimal policy choices when pollution is treated as a public good. Although his model also assumes no private demand for pollution cleaning services, it differs from Dufournaud et al. (1988) in two major ways. Firstly, the assumption of zero pollution discharge was relaxed to reflect empirical reality. Secondly, his model employed a Cobb-Douglas production function and a Stone-Geary utility function. To study the societal impacts of pollution and pollution abatement activities, he included both pollution and pollution-cleaning activities in the utility function. He also assumed that government pays for pollution cleaning services using revenue from Pigouvian taxes. Robinson (1990) mentions that the CGE model solution satisfies market equilibrium conditions but is not welfare maximising due to the existence of production externalities and the fact that pollution-cleaning is a public good. As a result, he constructed a nonlinear programming model (in which the CGE model equations serve as constraints in the program) to find fully optimal solutions. His results show that pollution control policies that work through changing prices and market incentives are more efficient. Other studies that also employed the fixed pollution coefficient approach include: Bergman (1991), Robinson et al. (1994), Goulder (1994), Smith and Espinosa (1996), Xie and Saltzman (2000), O'Ryan et al. (2005), Wen et al. (2010), and Fang et al. (2016). Moreover, theoretical environmental CGE models such as Rapanos (1992 and 1995) and Chua (2003) also specify pollution as a scalar multiple of output. Although intuitive, modelling pollution as fixed coefficients to sectoral output is restrictive because it does not

[3] Leontief incorporated pollution (measured in physical units) and a pollution removing activity into a conventional IO framework to show that environmental issues can be analyzed in an economy-wide framework.

allow polluters to introduce new technologies (such as substitution between inputs) that produce lower emissions.

As a result, the second group of environmental CGE models introduces substitution possibilities between inputs in production and/or utility functions. They assume a close connection between energy use and pollution generation and thus, allow substitution between energy (mainly fossil fuels) and non-energy inputs. In this manner, a tax on the use of fossil fuel (or on its carbon content) will induce polluters to substitute between composite energy (which is normally a CES aggregate of different energy inputs including fossil fuels) and other inputs to production. Predominantly, the models adopt a nested production structure and they focus on the economic impact of stabilizing greenhouse gas emissions. Bergman (1991) employed a CGE model that incorporated emissions and emissions control activities to analyse the general equilibrium effects of reducing NO_x^-, SO_x^- and CO_2 emissions in Sweden. The model distinguished between emissions from industrial processes and emissions from combustion. Emissions from combustion were assumed to be proportional to the use of fuel whereas industrial emissions were proportional to sectoral output. On the production side, he allowed substitution between composite energy input (which is a CES aggregation of electricity and fuels), capital, labour, and non-energy intermediate inputs using a CES aggregator function. He found that environmental policies are likely to have general equilibrium effects and that emission control cost functions that do not account for these effects may misrepresent the economic impact of emission control.

Conrad and Schröder (1993) analysed the economic impact of controlling four air pollutants (CO_2, SO_x, NO_x and particulates) on producers' as well as consumers' in the German state of Baden-Württenberg. They considered both production and consumption emissions and allowed substitution between energy and non-energy inputs/goods in the production and utility functions. Each sector combined energy (which is a CES aggregate of nine energy inputs), non-energy, labour, and capital to produce output using a CES specification. The demand side of the model differentiated between durable and non-durable goods with the assumption that the use of consumer durables (such as electric appliances and cars) contribute to air pollution through demand for fuel. They used the similarity between the dual concept in the theory of cost and production, and between the dual concept of utility and expenditure functions to model consumer behaviour. Their simulations showed that when emission taxes differ between producers and consumers, there is an inefficient allocation of resources in the economy. However, emission taxes are effective instruments for controlling air pollution.

Recently, Dissou (2005), and Oladosu and Rose (2007) also assumed substitution between energy and non-energy inputs to study the economic costs of reducing CO_2 emissions in Canada and the US, respectively. Dissou (2005) in his study assumed that CO_2 emissions emanate from the combustion of fossil fuels. The sectoral production structure followed a sequential decision process where the top level of the nest combines composite intermediate inputs, and a composite of value-added and energy. The second level of nesting combined aggregate labour and a composite of capital and energy using a Cobb-Douglas function. Aggregate energy input which is a CES specification of six fossil products is combined with capital using a CES function at the third level of nesting. The model was used to assess and compare the cost-effectiveness of a performance standard system and a permit-trading system to decrease CO_2 emissions in Canada. Results from the simulations suggest that the performance standard system could achieve the same level of productive efficiency as a market-based instrument because it introduces distortions in labour supply decisions. He, however, cautioned that the performance standard system should only be considered in some industries due to its potential high information and monitoring costs.

In addition to the above modelling approaches in terms of pollution generation, some studies treat the environment as a necessary input to production and/or consumption activities for which polluters require allowances to emit polluting substances. They then incorporate into the model a market for tradable emission permits with the cost of emission permits included in the production and/or utility functions. Brouwer et al. (2008) used a static, multi-sector (including a pollution abatement sector) model to study the general equilibrium impacts of protecting water quality in the Netherlands with focus on reducing eco-toxicological substances and emission levels of nutrients. Their model also included an emission permit market where the price of emission units is determined by equating supply and demand. The government supplies emission permits corresponding to the maximum allowed total emissions for each pollutant, and polluters (producers and consumers) demand emission permits in addition to other inputs and goods using a CES function. In this way, constraints on emissions enter directly into production and consumption decisions, and the price of emission permits is endogenously determined. They found that the total economic costs of reducing national emission levels vary between 0.2 and 9.4% of net national income depending on the stringency of the environmental policy in the base year. One novelty of their study is the downscaling of national and sector level results to the river basin level. At the sector and river basin level, they found that major pollution sources such as chemical and metal industry and commercial shipping in the largest river basin bear the largest share of the total economic costs. However, due to the static nature of their model, they could not study the flow of total annual economic adjustment costs of the emission reduction scenarios.

To address this limitation, Dellink et al. (2008) developed a dynamic version of their model to also study the economic cost of protecting water quality in the Netherlands. They found that the economic cost of national emission reduction is much lower in the dynamic model than in the static model. They attributed the differences in finding to the failure of the static model to represent the dynamic aspects of autonomous emission efficiency and developments in the abatement costs curves. Quin et al. (2011) employed the dynamic model developed by Dellink et al. (2008) to assess the economic impacts of water pollution mitigation measures in China. However, unlike the single abatement sector in Dellink et al. (2008), they disaggregated their abatement sector into three, each providing cleaning services for specific pollutants. Their simulations revealed that China can achieve a modest amount of emission reduction at low macroeconomic cost. Nonetheless, this cost will increase at a rate faster than linear as the stringency of policy targets increases. In addition, they found that stringent environmental policy could lead to changes in economic structure by shifting production away from dirty sectors. Bergman (1991) also included an emission permit market in his model with the government setting the environmental policy goals by restricting the number of permits.

Environmental CGE models also vary in the manner in which they model pollution abatement activities[4]. The differences lie in the specification of the pollution abatement cost function or pollution abatement technology. However, it is commonly assumed that economic incentives drive polluters' decision to engage in abatement activities by comparing the marginal cost of abatement to the price of emission permits or the uniform tax on pollution. Nestor and Pasurka (1995) explicitly modelled the pollution abatement technology of German industries using a dataset in which the specific inputs used for environmental protection are in an IO matrix. They showed that an accurate specification of abatement costs is important in assessing the economic costs of environmental policy. Yet, other studies employ simplifying assumptions about abatement technology due to data limitations. Robinson et al. (1994) used a Leontief production function to model pollution abatement and

[4] Pollution abatement is any measure that reduces the environmental pressure of economic activity (i.e. it reduces the amount of pollution generated per unit of production or consumption). The cost incurred in the process is the pollution abatement cost.

assumed that the marginal cost of abatement equals the unit cost of inputs used by the abatement sector. Conrad and Schröder (1993) also employed a Leontief abatement technology to model abatement activities. Hazilla and Kopp (1990), and Jorgenson and Wilcoxen (1990) assumed that the technology used in the abatement sector mirrors that in the production sector. Robinson (1990) and Xie and Saltzman (2000) modelled pollution abatement using a Cobb-Douglas production function.

Beyond the differences in modelling pollution and abatement activity, some studies consider feedback effects such as the impact of environmental degradation on labour productivity and on the welfare of consumers. These effects are incorporated in the production and/or utility functions. Piggott et al. (1992) incorporated the environmental benefits of reduced carbon emissions in the utility function of their model to study the international impacts of carbon reduction initiatives. Xie and Saltzman (2000) also included in their social welfare function, the level of pollution abatement to reflect the effect of pollution cleaning on social welfare. Bergman (1995) incorporated his environmental quality index in both the utility and production functions whereas Gruver and Zeager (1994) included theirs in the production function.

Another strand of environmental CGE models that deserve mentioning is the models that econometrically estimate their behavioural parameters. These models are mostly intertemporal models that focus on the cost of environmental regulation without considering the benefits. Studies in this category include Hazilla and Kopp (1990) and Jorgenson and Wilcoxen (1990), both of which follow the pioneering work of Hudson and Jorgenson (1974). Hazilla and Kopp (1990) studied the social costs of environmental quality regulations mandated by the clean air and water acts. Their measure of social cost was based on household willingness to pay rather than the expenditures they incur in complying with the new policy. They simulated two scenarios (with and without the regulation) to estimate the social costs of the clean air and water programs. The results show that the social cost estimates of environmental regulation sharply diverge from the private cost. Jorgenson and Wilcoxen (1990) also simulated U.S. economic growth in the long-term with and without pollution control. They found that pollution abatement places a major demand on the resources of the U.S economy.

In South Africa, the CGE approach has also been extensively used to analyse the economic costs and effectiveness of environmental management policies. Several studies have employed the framework to analyse the economic and environmental impacts of a greenhouse gas mitigation policy in SA (Pauw, 2007; Devarajan *et al.,* 2011; Alton *et al.,* 2014; Van Heerden *et al.,* 2016). Regarding water management, studies have analysed the impact of water policies on water use and allocation as well as the associated costs and trade-offs (Hassan & Thurlow, 2011). Other studies have focused on the impacts of water pricing (Lestoalo *et al.,* 2007; Van Heerden *et al.,* 2008; Gill & Punt, 2010), climate change (Juana *et al.,* 2008), and macroeconomic policies (Blignaut & Van Heerden, 2009) on sectoral water allocations, the environment, household welfare and economic growth.

The preceding literature review shows the extensive application of CGE models in the analysis of environmental policy issues. Nonetheless, it also highlights the limited application (Xie and Saltzman, 2000; Brouwer et al., 2007; Dellink et al., 2008; Wen et al., 2010; Quin et al., 2011; Fang et al., 2016) of the approach in the analysis of water quality issues. Moreover, CGE models in SA (Mukherjee, 1996; Lestoalo et al., 2005; DEA, 2007; Juana et al., 2008; Van Heerden et al., 2008; Blignaut & Van Heerden, 2009; WRC, 2008; Gill & Punt, 2010; Hassan & Thurlow, 2011) have so far focused on managing the quantity dimension of water, and to the best of the authors knowledge no effort has so far been made to analyse water quality management dimensions using the CGE framework. Accordingly, this study attempts to contribute to bridging this gap in the literature by adapting a CGE model to include information on water pollution and abatement measures to analyse the basin-wide impacts and effectiveness of water quality management policies in the Olifants river basin of South Africa.

For this purpose, the IFPRI standard CGE model is adapted to the requirements of a regional model and to include a production function for pollution abatement activities that have the responsibility of providing the best available cleaning services to help polluters meet prescribed environmental standards. The 'output' of pollution abatement activities is treated as a special intermediate good bought by polluters. The amount paid by polluters for these special intermediate goods constitutes their abatement cost. It is also assumed that not all pollution generated in the economy can be removed by the abatement sectors, so the government levies a tax on the amount not removed (unabated pollution). Therefore, the cost of pollution control which includes water pollution tax and abatement cost is included in the cost of production (i.e., based on the 'polluter pays' principle). The next chapter discusses in detail the analytical framework and the implementation of the regional environmental CGE model.

2.4 **Summary**

This chapter reviews the relevant literature on analytical approaches and empirical methods employed to study the social and economic implications of environmental policies and related studies. The chapter reveals that inasmuch as the implementation of environmental policies induces both direct and indirect costs, a proper assessment of their economic and social costs requires an integrated approach. This approach integrates the direct cost which is the abatement cost incurred by polluters' to meet the prescribed policy targets with the indirect costs which involves the forgone profits and utility as a result of agents' behavioural changes in response to the policy. The indirect costs can only be properly assessed in an economy-wide multi-sector model. Within the class of economy-wide modelling approaches, the review shows that the CGE approach with its appeal of endogenous price determination and a substitution possibility in supply and demand systems is capable of more accurately simulating the results of environmental policy changes than the other approaches.

CHAPTER 3: ANALYTICAL FRAMEWORK AND EMPIRICAL METHODS

3.1 Introduction

This chapter is divided into two parts. The first part which is the analytical framework begins with a brief overview of the interaction between an economy and its environment. Following an overview of the IFPRI standard CGE model in section 3.3, the modifications and additions made to the generic model to address the objectives of this study are presented in section 3.4.

The second part is the empirical methods and it begins with a description of the case study area. Section 3.6 provides a brief introduction to the framework of an environmentally extended Social Accounting Matrix (ESAM). An ESAM integrates both economic and environmental information and is the commonly employed data structure for calibrating environmental CGE models. Section 3.7 outlines the sources of economic and environmental data used for the construction of the Olifants ESAM. Section 3.8 motivates the approach taken to estimate the parameters of the Olifants environmental CGE model by comparing two parameter estimation approaches. Section 3.9 provides a summary of the chapter.

3.2 The Interactions between an Economy and its Environment

The interactions between an economy and its environment (including water and air) occur in complex and diverse ways. The economic system is concerned with the production and consumption of goods and services which are made possible by the material resources and energy provided by the environment. The economy also produces waste which is discharged back into the environment. For instance, raw water supplied by the Olifants River is used as an input in the economic production processes of the various economic sectors in the basin. The river also serves as a sink for a variety of effluents and by-products from the economic production processes. However, the capacity of the river/environment to assimilate waste from the economy and supply material resources is limited. This limited environmental capacity constraints socioeconomic growth and development. The environment also provides amenity services (such as water sports and other sources of stimulation and pleasure provided by the biosphere) which flow directly into utility. Figure one is a schematic representation of the interactions between an economy and its environment.

The diagram shows that the environment provides three services to the economic system – resource supply, direct source of utility (i.e. amenity services), and waste assimilation. When the amount of waste (W) flowing from the economic system is in excess of the environment's assimilative capacity (A), environmental degradation occurs, negatively impacting utility and the stock of environmental resource.

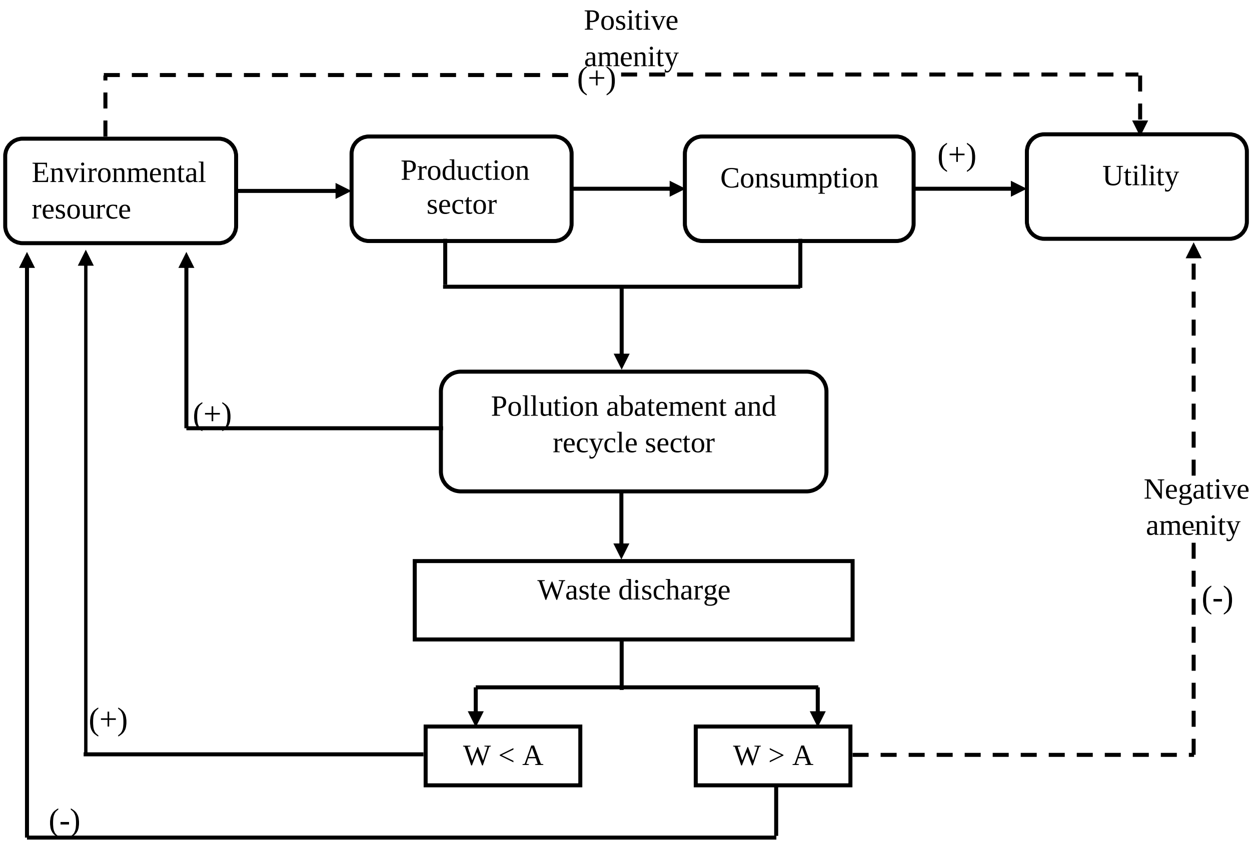


Figure 3.1: Interactions between the Economy and the Environment. Adapted from Pearce and Turner (1990)

For example, the polluted state of the Olifants River shows that its assimilative capacity has been exceeded due to the effluent discharged by the industries and poorly functioning sewage treatment plants in the basin. As a result, the capacity of the river to support aquatic life and recreational activities is threatened. The pollution abatement and recycling sector, on the other hand, reduces the amount of waste discharged per unit of economic activity and treats effluents prior to being discharged into the river. Thus, the sector helps to reduce the demand on the environment's assimilative capacity and the environmental resource base. Nonetheless, polluters have no incentive to account for these costs (due to the externality nature of pollution damage) unless there are regulations to force them to do so.

The government thus needs to choose and implement effective pollution control or environmental protection policies to reduce waste/effluents and protect the environment. Notable pollution control policies include environmental standards, tradable emission permits, pollution taxes, and pollution abatement subsidies. Environmental standards work by setting uniform standards for economic agents regardless of the relative cost to them of achieving the standards. The standards could be based on technology (i.e. the regulator prescribes the method and sometimes the actual equipment that the polluter must use) or performance (i.e. the polluter has the freedom to choose how to meet the standard) and are monitored by the regulator to ensure compliance. Failure by polluters to comply with the set standards results in a penalty. The South African water quality guideline for aquatic ecosystems is an example of a performance standard. On the contrary, pollution taxes, tradable emission permits, and pollution abatement subsidies are classified as economic or market-based instruments because they operate by changing the behaviour of economic agents through market signals rather than through mandatory restrictions. These instruments are favoured by economists because they achieve policy targets at least-cost. A pollution emission tax is a price on emission which is usually based on the external damage/cost of the emission. An example in SA is the Waste Discharge Charge System (WDCS). With the subsidy approach, the regulator provides financial incentives for the installation of pollution abatement equipment to reduce pollution. The tradable emission permits system is similar to standards in that the regulator allows only a certain level of pollution emissions. However, unlike standards, the regulator issues permits for the allowed level of pollution, which can be bought and sold in a permit market. The main purpose of this study is to adapt a CGE model to include water pollution and abatement measures to assess the economic, environmental, and distributional impacts of WQM policies such as pollution taxes and subsidies in the Olifants river basin.

3.3 **Overview and Equations of the IFPRI Standard CGE Model**

The preceding section motivates the need for an analytical framework that integrates both economic and environmental activities in order to achieve the purpose of this study. Moreover, the approach should account for both the direct costs (such as compliance cost to the polluter) and indirect costs associated with the environmental policy (such as forgone utility and profit by economic agents' as a result of behavioural changes in response to the policy). In this regard, this section presents the economic part of such an integrated model while the next section presents the environmental component.

We adapt the IFPRI standard static CGE model specifications of Löfgren et al. (2002) to the requirements of a regional[5] model. The model specifications follow the neoclassical-structuralist modelling tradition pioneered by Dervis et al. (1982) and others. The version presented here; however, exclude certain features of the generic model, due to data limitations. The features omitted include home consumption of domestically produced goods, transaction costs for commodities that enter the market space, and the assumption that activities produce multiple products. It is important, however, to know that the omission of these features does not impact on the validity of our model. The model makes a distinction between activities (i.e. entities that carry out production) and commodities (i.e. the output of activities). By way of notational principle, the model adopts (1) upper case letters for endogenous variables, (2) upper case letters with a bar or lower-case letters for exogenous variables and (3) Greek letters for parameters. Indices appear as subscripts in lower case letters to variables and parameters, and consist of activities (a), commodities (c), factors of production (f), and institutions (i including households (h), enterprise and rest of the world (ROW)). Commodity prices start with p, factor prices with w, and commodity and factor quantities with q. In the presentation that follows, the model equations are divided into four blocks: price, production and trade, institutional incomes and expenditures, and system constraints.

3.3.1 Price Block

The model has a detailed price system, mainly because of the accepted quality differences among commodities of different origins and destinations. There are a total of seven endogenous prices in the model: the domestic prices of imports and exports (PM and PE), composite commodity price (PQ), average producer price (PX), gross activity price (PA), aggregate intermediate input price (PINTA), and the price of value-added (PVA). Table 3.1 shows how these prices are linked to other exogenous prices and nonprice variables.

Given that it is not possible to estimate changes in relative prices and economic output in other countries, the model assumes that SA is a small open economy. As a result, the basin's export and import prices are determined by the world market. In equation (1), the domestic price of imported commodities (PM) is determined by the world price (pwm) inclusive of tariff (tm) times the exchange rate (EXR) (local currency per unit of foreign currency). Domestic producers that export their output receive a price (PE) that is determined by the

[5] This involved the construction of a regional SAM and adopting new closure rules for the accounts of the government and the rest of the world. We use region and basin interchangeably.

world price of exports (pwe) exclusive of tax (te) times the exchange rate (Equation 2). A distinction is made between composite commodities QQ (i.e. goods available for domestic consumption) and commodities produced by domestic activities (QX).

Table 3.1: Price Equations

(1)	$PM_c = pwm_c.(1 + tm_c).EXR$	Import price in domestic currency
(2)	$PE_c = pwe_c(1 - te_c).EXR$	Export price in domestic currency
(3)	$PQ_c = \dfrac{PD_c.QD_c + PM_c.QM_c}{QQ_c}(1 - tq_c)$	Average demand price of composite commodity
(4)	$PX_c = \dfrac{PD_c.QD_c + PE_c.QE_c}{QX_c}$	Average producer price of commodity c
(5)	$PA_a = \sum_c PX_c.\theta_{a\,c}$	Gross activity price
(6)	$PINTA_a = \sum_c PQ_c.ica_{c\,a}$	Aggregate intermediate input price
(7)	$PA_a.(1 - ta_a).QA_a = PVA_a.QVA_a + PINTA_a.QINTA_a$	Activity revenue and cost (this equation implicitly defines the price of value-added, PVA)
(8)	$\overline{CPI} = \sum_c PQ_c.cwts_c$	Consumer price index (numeraire)

Equation (3) describes the price (PQ) paid by domestic demanders for the composite good which is a CES aggregation of sectoral imports (QM) and domestic commodities supplied to the domestic market (QD). Equation (4) determines the price (PX) for sectoral output which is a Constant Elasticity of Transformation (CET) aggregation of domestic supply (QD) and export supply (QE). Equation (5) defines the activity price (PA) as yields per activity unit ($\theta_{a\,c}$) multiplied by activity-specific commodity prices. Activity-specific aggregate intermediate input price (PINTA) is defined as the product of composite commodity prices and intermediate input coefficients ($ica_{c\,a}$) (Equation 6). Total revenue for each sector (exclusive of taxes) is split into the cost of primary factors (capital and labour) and intermediate input cost (Equation 7). This equation will be modified to include the cost of pollution control in the next section. Equation (8) defines the model's numéraire against which all relative prices are compared.

3.3.2 Production and Trade Block

The model assumes that all producers (each represented by an activity) maximise profit subject to a production technology. The production technology follows a nested structure with the understanding that substitution varies between inputs. Also, the model assumes imperfect substitutability between domestic and traded commodities using CES type functions. Equations defining production and trade are presented in Table 3.2

Table 3.2: Production and Trade Equations

(9)	$QVA_a = iva_a . QA_a$	Demand for aggregate value-added
(10)	$QINTA_a = inta_a . QA_a$	Demand for aggregate intermediate input
(11)	$QINT_{c\,a} = ica_{c\,a} . QINTA_a$	Demand for disaggregated intermediate input
(12)	$QVA_a = \alpha_a^{va} . \left(\sum_{f \in F} \delta_{f\,a}^{va} . QF_{f\,a}^{-\rho_a^{va}} \right)^{-\frac{1}{\rho_a^{va}}}$	Aggregate value-added
(13)	$WF_f . \overline{WFDIST}_{f\,a} = PVA_a . QVA_a . \left(\sum_{f \in F} \delta_{f\,a}^{va} . QF_{f\,a}^{-\rho_a^{va}} \right)^{-1} . \delta_{f\,a}^{va} . QF_{f\,a}^{-\rho_a^{va}-1}$	Demand for factor f from activity a
(14)	$QX_c = \alpha_c^t . \left(\delta_c^t . QE_c^{\rho_c^t} + (1 - \delta_c^t) . QD_c^{\rho_c^t} \right)^{\frac{1}{\rho_c^t}}$	Output transformation function
(15)	$\frac{QE_c}{QD_c} = \left(\frac{PE_c}{PD_c} . \frac{1 - \delta_c^t}{\delta_c^t} \right)^{\frac{1}{\rho_c^t - 1}}$	Export–domestic supply ratio
(16)	$QX_c = QD_c + QE_c$	Output transformation for non-exported commodities
(17)	$QQ_c = \alpha_c^q . \left(\delta_c^q . QM_c^{-\rho_c^q} + \left(1 - \delta_c^q\right) . QD_c^{-\rho_c^q} \right)^{-\frac{1}{\rho_c^q}}$	Composite supply function
(18)	$\frac{QM_c}{QD_c} = \left(\frac{PD_c}{PM_c} . \frac{\delta_c^q}{1 - \delta_c^q} \right)^{\frac{1}{1+\rho_c^q}}$	Import–domestic demand ratio
(19)	$QQ_c = QD_c + QM_c$	Composite supply for non-imported outputs and non-produced imports

At the top level of the production technology, aggregate value-added (QVA) and aggregate intermediate input (QINTA) are defined as Leontief functions of activity level (Equations 9 and 10). Demand for disaggregated intermediate inputs (QINT) is defined as the level of aggregate intermediate input use times a fixed intermediate input coefficient (Equation 11). Aggregate value-added which is a combination of different labour skills (unskilled, skilled and highly skilled) and capital is modelled using a CES aggregation function (Equation 12). Imported goods are assumed to be imperfect substitutes in production and consumption thus; aggregate intermediate input is a CES Armington function of domestic and foreign goods. Given this structure, producers aim to maximise profit from production (QA) taking the prices for factors, intermediate inputs, and output as given. Demand for production factors (QF) excluding intermediate inputs is obtained by differentiating equation (12). Production factors are employed up to the point where the marginal revenue product of each factor equals its price (WF) (Equation 13). The model incorporates a distortion variable (WFDIST) to reflect the fact that factor prices may differ across sectors due to their mobility or other considerations. Thus, the distortion variable measures how far a sector's factor price is from the economy-wide average (WF). Producer's decision to export or sell in the domestic market is based on the CET aggregation function (Equation 14). Domestic demand for the composite good is also governed by the CES aggregation function (Armington function) (Equation 17). Equations (15 and 18) respectively define the export supply and import demand functions, both of which depend on relative prices. Equation (16) replaces equation (15) for commodities for which total output is allocated to either the domestic market or to the export market. Also, equation (18) is replaced by equation (19) for domestic commodities with no imports and for imports with no domestic production.

3.3.3 Institutional Incomes and Expenditures Block

Institutions in this regional CGE model are represented by households, government, enterprise, and the ROW[6]. Consumption decisions by households are driven by the maximization of utility. Government consumption is assumed to be fixed in real terms whereas the enterprise sector is assumed not to consume. Factor incomes accrue to households and enterprise whereas tax revenues accrue to the government. The flow of income to institutions and their expenditure are presented in Table 3.3.

[6] ROW in this case includes all other regions in SA (i.e., outside study region), in addition to foreign countries.

Table 3.3: Institutional Income and Expenditure Equations

(20)	$YF_f = \sum_a WF_f . \overline{WFDIST}_{f\,a} . QF_{f\,a}$	Income of factor f
(21)	$YIF_{i\,f} = shif_{i\,f} . \left[\left(1 - tf_f\right) . YF_f - trnsfr_{row\,f} . EXR \right]$	Income of domestic institution i from factor f
(22)	$YH_h = \sum_f YIF_{h\,f} + trnsfr_{h\,gov} . \overline{CPI} + trnsfr_{h\,row} . EXR$	Household income
(23)	$HHSAV_h = mps_h . (1 - ty_h) . YH_h$	Household savings
(24)	$EH_h = (1 - ty_h) . YH_h - trnsfr_{h\,h} - HHSAV_h$	Household consumption expenditure
(25)	$PQ_c . QH_{c\,h} = PQ_c . \gamma^m_{c\,h} + \beta^m_{c\,h} \left(EH_h - \sum_c PQ_c . \gamma^m_{c\,h} \right)$	Household consumption demand
(26)	$QINV_c = \overline{IADJ} . \overline{qinv}_c$	Investment demand
(27)	$YG = \sum_h ty_h . YH_h + EXR . trnsfr_{gov\,row} + \sum_a ta_a . PA_a . QA_a + \sum_c tm_c . pwm_c . QM_c . EXR + \sum_c te_c . pwe_c . QE_c . EXR + \sum_c tq_c . PQ_c . QQ_c$	Government income
(28)	$EG = \sum_c PQ_c . \overline{QG}_c + \sum_h trnsfr_{h\,gov} . \overline{CPI}$	Government expenditure

Total income for each production factor is determined by the sum of its demand across activities (Equation 20). Factor income is distributed among domestic institutions in fixed shares ($shif_{i\,f}$) after transfers to the rest of the world (Equation 21). Household income is the sum of factor income and transfers from other institutions such as government and rest of the world (Equation 22). They then pay income tax (ty_h) and save a portion of their income based on fixed saving propensities (MPS_h) (Equations 23). The rest of the income (EH) is spent on consumption according to the Linear Expenditure System (LES) of demand (Equations 24 and 25). Fixed investment demand is defined as a product of base-year quantity and an exogenous adjustment factor (Equation 26). Government revenue (YG) derives from various tax accounts (including tariff, income tax, and production tax) and transfers from the rest of the world (Equation 27). This equation will be modified to include government revenue from water pollution taxes in the next section. Equation (28) defines government expenditure (EG) as the sum of government spending on consumption (QG) and transfers to other institutions. Government consumption is assumed to be fixed in real terms whereas its saving is determined residually.

3.3.4 System Constraint Block

The model has a set of equilibrium conditions that must be satisfied by the system as a whole but not necessarily considered by any individual actor. They include factor and commodity markets, government balance, savings, and investment balance, and the current account of the rest of the world. Each equilibrium condition is governed by a set of closure rules that ensures that there is a balance between the number of endogenous variables and independent equations in the system (Löfgren, 1995). Nonetheless, it is difficult to argue for a single rule thus, we examine in our policy simulations' in chapters four and five alternative closure scenarios. These equations are presented in Table 3.4.

Table 3.4: System Constraint Equations

(29)	$\sum_{a} QF_{f\,a} = \overline{QFS}_f$	Market equilibrium for factors
(30)	$QQ_c = \sum_{a} QINT_{c\,a} + \sum_{h} QH_{c\,h} + \overline{QG}_c + QINV_c$	Market equilibrium for composite commodity
(31)	$\sum_{c} pwm_c . QM_c + \sum_{f} trnsfr_{row\,f} = \sum_{c} pwe_c . QE_c + \sum_{i} trnsfr_{i\,row} + \overline{FSAV}$	Current account balance (in foreign currency)
(32)	$GSAV = YG - EG$	Government balance
(33)	$\sum_{h} HHSAV_h + GSAV + EXR.\overline{FSAV} = \sum_{c} PQ_c . QINV_c$	Savings-investment balance

Equation (29) defines the market clearing condition for factors with average factor prices adjusting to achieve equilibrium. Equilibrium in the commodity market requires that sectoral supply equals demand (Equation 30). Equations (31 and 33) respectively define the current account balance and the equality between aggregate savings and aggregate investment. Foreign savings is exogenously determined thus, in equation (33) the equilibrating variable is the exchange rate. Government savings is determined residually as the difference between government revenue and spending (Equation 32).

3.4 The Environmental Component

The environmental component of this regional CGE model includes information on water pollution and abatement activities of production sectors, pollution taxes, and subsidies. That is, the environment in this study is captured through water pollution and abatement. The modelling approach follows that of Xie & Saltzman (2000) however, we exclude certain features such as consumption pollution because of data limitations, and since production activities constitute the major source of pollution in the OWMA (DWA, 2011c). We modify a number of equations in the generic CGE model to reflect pollution-related costs incurred by polluters as well as government revenue from water pollution taxes. Also, we specify equations describing the cost of pollution control and the total amount of pollution abated and emitted in the economy.

The production set of the generic model is extended to include pollutants and pollution abatement activities. We include a production function for pollution abatement activities and treat their 'output' as special intermediate goods bought by polluters. The incorporated pollution abatement sectors have the responsibility of providing the best available cleaning or purification services to help polluters meet prescribed environmental standards. The amount paid by polluters for these special intermediate goods constitutes their abatement cost. The demand and price of pollution abatement services are endogenously determined in the model based on prevailing market conditions. Our model also assumes that not all pollution generated in the economy can be removed by the abatement sectors, so the government levies a tax on the amount not removed (unabated pollution). Therefore, the cost of pollution control which includes water pollution tax and abatement cost is included in the cost of production (i.e., based on the 'polluter pays' principle). That is, pollution is linked to the total production of each sector on the input side of the production function. In this manner, the direct cost of pollution control is integrated with a top-down model that determines the indirect costs.

Next, we present the modified equations as well as the new ones related to pollution control. Pollution abatement activities and pollutants are indexed with g and it appears as a subscript to variables and parameters. Table 3.5 presents the pollution related equations.

Table 3.5: Pollution Equations

7 modified	$PA_a.(1-ta_a).QA_a = PVA_a.QVA_a + PINTA_a.QINTA_a + \sum_g PACOST_{g\,a} + \sum_g PETAX_{g\,a}$	Activity revenue and cost (including the cost of pollution control)
27 modified	$YG = \sum_h ty_h.YH_h + EXR.trnsfr_{gov\,row} + \sum_a ta_a.PA_a.QA_a + \sum_c tm_c.pwm_c.QM_c.EXR + \sum_c te_c.pwe_c.QE_c.EXR + \sum_c tq_c.PQ_c.QQ_c + \sum_g \sum_a PETAX_{g\,a}$	Government income (including income from water pollution taxes)
34	$PAG_g = \left(\frac{QA0_g}{TDA0_g}\right).PA_g$	Pollutant abatement price conversion
35	$PETAX_{g\,a} = tpe_g.d_{g\,a}.QA_a.(1-cl_g)$	Pollution emission tax
36	$PACOST_{g\,a} = PAG_g.d_{g\,a}.QA_a.cl_g$	Pollution abatement cost
37	$TDA_g = QA_g.\left(\frac{TDA0_g}{QA0_g}\right)$	Total pollution abated
38	$DG_g = \sum_a d_{g\,a}.QA_a$	Total pollution generated
39	$DE_g = DG_g - TDA_g$	Total pollution emitted

Equation (7) now includes the cost of pollution control to the polluter i.e., the cost of pollution abatement ($PACOST_{g\,a}$), and pollution emission tax ($PETAX_{g\,a}$). Pollution emission tax and pollution abatement cost are defined in equations (35 and 36), respectively. It should be noted that equation (7) modified defines the price of pollution abatement services in the same way as the price of conventional products (i.e. the amount of one-rand worth of product in the base year). However, it is more expedient to measure pollution in tonnes thus; equation (34) converts the price of pollution clean-up (PA_g) to rand per tonne using base year values of pollution abatement ($\mathrm{QA0_g}$ and $\mathrm{TDA0_g}$). Equation (27) also now reflects the amount of revenue raised by the government from taxes levied on production pollution emissions.

Equation (35) defines pollution emission tax by pollutant and by activity ($PETAX_{g\,a}$) as the product between the policy determined tax rate ($\mathrm{tpe_g}$) and the total level of pollution emission ($\mathrm{d_{g\,a}}.\mathrm{QA}_a.(1-\mathrm{cl}_g)$). cl_g is the average production pollution clean-up rate thus, $(1-\mathrm{cl}_g)$ is the proportion of pollutant g discharged into the environment, $\mathrm{d_{g\,a}}$ is pollution coefficient per unit of activity output (with unit tonnes of pollutants per unit of one-rand worth output) and QA_a is activity output. The clean-up rate represents the idea that polluters' are able to remove a certain proportion of each pollutant by purchasing abatement commodities or investing in abatement technologies that reduces pollution per unit of output. Equation (36) determines pollution abatement cost by pollutant and by activity ($\mathrm{PACOST_{g\,a}}$) as a function of the price of pollution clean-up services ($\mathrm{PAG_g}$) and amount of pollutant abated ($\mathrm{d_{g\,a}}.\mathrm{QA}_a.\mathrm{cl_g}$). It is important to note that equations (35) and (36) implicitly determine the quatities of pollution emitted and abated, respectively. Equation (37) converts the output of the pollution abatement sector ($\mathrm{QA_g}$) (determined in the production and trade block) to the level of total pollution abatement measured in tonnes ($\mathrm{TDA_g}$) using base year values of both variables ($\mathrm{TDA0_g}$ and $\mathrm{QA0_g}$). Total pollution generated (DG_g) and total pollution emitted (DE_g) is determined by equations (38) and (39), respectively. Total pollution generated is the product of pollution coefficient and activity output whereas total pollution emitted is the difference between total pollution generated and the level of pollution abatement. The equilibrium conditions defined under the system constraint block include equilibrium of pollution abatement activities and their demand for intermediate inputs and primary factors. Pollution clean-up services are however, assumed to be non-tradable so their trade-related variables are set to zero. Table 3.6 presents a dictionary of set, parameters and variable names.

Table 3.6: Model Set, Parameters and Variables

SETS	
$a \in A$	Activities
$c \in C$	Commodities
$f \in F$	Factors
$i \in INS$	Institutions (including households, enterprise, government and rest of the world)

$h \in H (\subset INS)$	Households
$g \in IA$	Pollutants/pollution abatement activities
PARAMETERS	
Latin letters	
cwts_c	Weight of commodity c in the CPI
cl_g	Cleanup rate (maximum reduction) of pollutant g produced from activity a (unitless)
$\mathrm{ica}_{c\,a}$	Quantity of commodity c as intermediate input per unit of activity a
inta_c	Quantity of aggregate intermediate input per activity unit
iva_a	Quantity of value-added per activity a
pwe_c	Export price (foreign currency)
pwm_c	Import price (foreign currency)
$\mathrm{shif}_{i\,f}$	Share for domestic institution i in income of factor f
ta_a	Tax rate for activity a
te_c	Export tax rate
tf_f	Direct tax rate for factor f
tm_c	Import tariff rate
tpe_g	Water pollution tax rate
tq_c	Rate of sales tax
ty_h	Rate of income tax for household h
Greek letters	
α_c^q	Armington function shift parameter
α_c^t	CET function shift parameter
α_a^{va}	Efficiency parameter in the CES value-added function
$\beta_{c\,h}^m$	Marginal share of consumption spending on commodity c for household h
$\gamma_{c\,h}^m$	Subsistence consumption of commodity c for household h
δ_c^q	Armington function share parameter
δ_c^t	CET function share parameter
$\delta_{f\,a}^{va}$	CES value-added function share parameter for factor f in activity a
$\theta_{a\,c}$	Yield of output c per unit of activity a
ρ_c^q	Armington function exponent
ρ_c^t	CET function exponent
ρ_a^{va}	CES value-added function exponent
ENDOGENOUS VARIABLES	
DE_g	Total amount of pollutant g emitted
DG_g	Total amount of pollutant g generated
EG	Government expenditure
EH_h	Consumption spending for household h
EXR	Exchange rate (local currency unit per unit of foreign currency unit)

GSAV	Government savings
$\mathrm{HHSAV_h}$	Household savings
PA_a	Activity price
$\mathrm{PACOST_{g\,a}}$	Pollution abatement cost
$\mathrm{PAG_g}$	Price of pollutant g abated
$\mathrm{PD_c}$	Domestic price for commodity c
$\mathrm{PE_c}$	Export price (domestic currency)
$\mathrm{PETAX_{g\,a}}$	Pollution emission tax
$\mathrm{PINTA_a}$	Aggregate intermediate input price for activity a
$\mathrm{PM_c}$	Import price (domestic currency)
$\mathrm{PQ_c}$	Composite commodity price
$\mathrm{PVA_a}$	Value-added price (factor income per unit of activity)
$\mathrm{PX_c}$	Aggregate producer price for commodity c
$\mathrm{QA_a}$	Quantity (level) of activity
$\mathrm{QD_c}$	Quantity sold domestically of domestic output
$\mathrm{QE_c}$	Quantity of exports
$\mathrm{QF_{f\,a}}$	Quantity demanded of factor f from activity a
$\mathrm{QH_{c\,h}}$	Quantity consumed of commodity c by household h
$\mathrm{QINTA_a}$	Quantity of aggregate intermediate input
$\mathrm{QINT_{c\,a}}$	Quantity of commodity c as intermediate input to activity a
$\mathrm{QINV_c}$	Quantity of investment demand for commodity c
$\mathrm{QM_c}$	Quantity of imports of commodity c
$\mathrm{QQ_c}$	Quantity of goods supplied to domestic market (composite supply)
$\mathrm{QVA_a}$	Quantity of aggregate value-added
$\mathrm{QX_c}$	Quantity of domestic output of commodity c
$\mathrm{TDA_g}$	Total amount of pollutant g abated
$\mathrm{WF_f}$	Average price of factor f
$\mathrm{YF_f}$	Income of factor f
$YIF_{i\,f}$	Income to domestic institution i from factor f
$\mathrm{YH_h}$	Household income
YG	Government revenue
EXOGENOUS VARIABLES	
$\overline{CPI}$	Consumer price index
$\overline{FSAV}$	Foreign savings (foreign currency unit)
$\overline{GADJ}$	Government consumption adjustment factor
$\overline{IADJ}$	Investment adjustment factor
$\overline{QFS}_f$	Quantity supplied of factor f
$\overline{WFDIST}_{f\,a}$	Wage distortion factor for factor f in activity a
$\overline{QG}_c$	Government consumption demand for

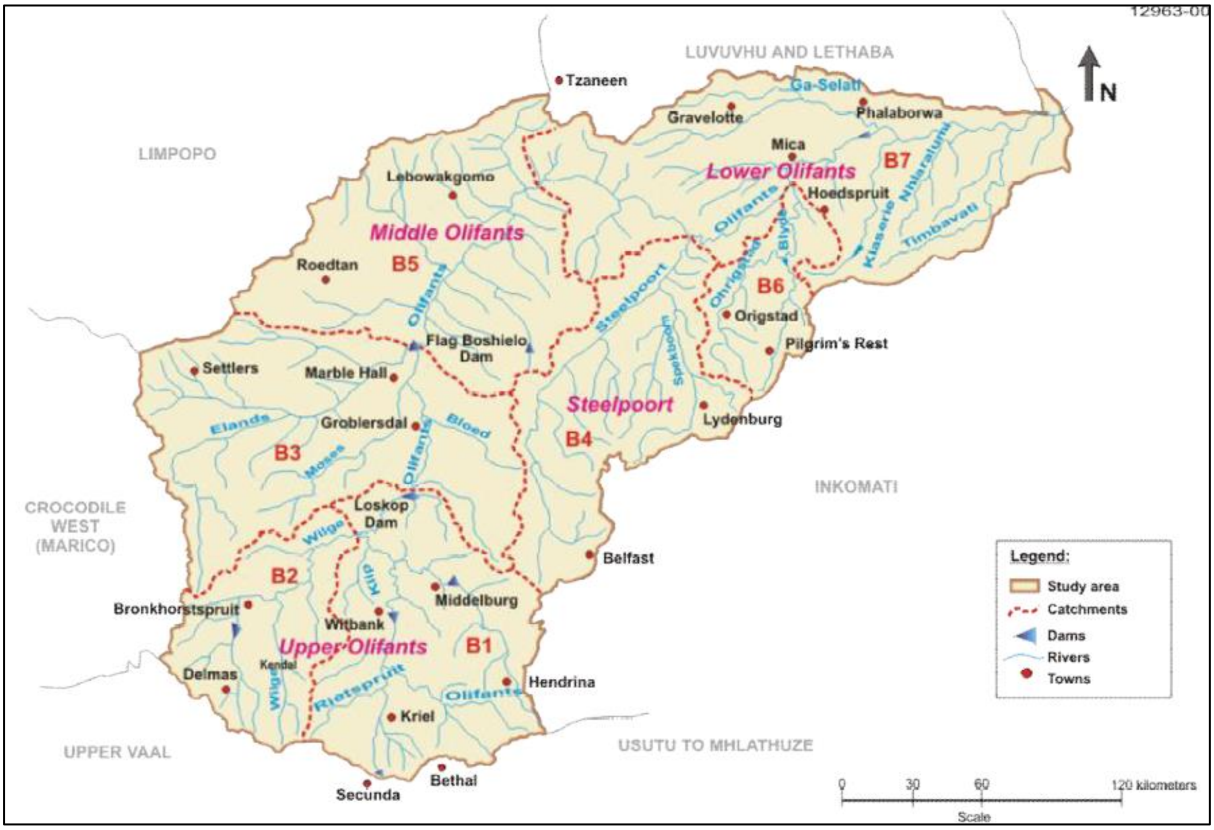
	commodity c
$\overline{\text{qinv}}_c$	Base-year quantity of investment demand
$\text{trnsfr}_{i\,f}$	Transfer from factor f to institution i
$\text{trnsfr}_{i\,i}$	Transfer from institution i to i

The aim of the first part of this chapter was to present the technical specification of the regional environmental CGE model (i.e. a model that integrates economic activities and water pollution related activities). However, for the model to be useful for environmental policy analysis, the appropriate environmental and economic data, as well as the correct specification of the model parameters are required. We turn to these issues in the second part of this chapter by beginning with a description of the case study area.

3.5 **The Case Study Area**

The Olifants River basin is located in the north-eastern part of South Africa and straddles three provinces – Limpopo, Mpumalanga, and Gauteng provinces. The catchment covers an area of about 54 570 km^2 with a total mean annual runoff of about 2400 million m^3/annum and has varying climatic conditions. Hydrologically, the Olifants River has a dense network of tributaries particularly the Klein Olifants River, Elands River, and the Wilge River. The flow of the tributaries has been interrupted by abstraction and the main stem of the river is highly regulated with many dams interspersed along its course. There are 37 major and 134 minor dams, all classified and registered by the Department of Water and Sanitation (UNEP, 2015). The climate and the dense network of rivers give rise to diverse flora and fauna, some only found in the Olifants.

From the water management viewpoint, the Olifants catchment is divided into four sub-areas namely the Upper Olifants, Middle Olifants, Lower Olifants, and Steelpoort areas (see Figure 2). The varying economic activity in these areas has led to enormous water demand and deterioration of water quality in the catchment. For instance, the Upper Olifants sub-area supports extensive mining and coal-fired power generation which support 48% of SA's total power generation capacity (UNEP, 2015). As a result, the Upper Olifants River is characterised by extensive pollution and acidification as a result of mining and industrial activities. The middle section of the catchment is characterised by agricultural activities (both commercial and subsistence), human settlements, and is home to the largest irrigation scheme in the country. Given the importance of the Olifants River to economic activities in the OWMA, there is a need to manage the water resource in a manner that ensures an equitable balance among the multiple users and uses. In this context, the OWMA was selected as a case study area with the objective of analysing the economic implications of protecting the water resources in the catchment.



Source: DWS (2011b)

Figure 3.2: The Olifants Water Management Area and the Four Sub-Areas

3.6 Framework for an Environmental Social Accounting Matrix

The concept of a Social Accounting Matrix (SAM) as an accounting framework and its use for calibrating CGE models and as a tool for policy analysis has been reviewed in chapter 2. As pointed out by Bartelmus et al. (1993), a conventional SAM fails to represent the degradation of environmental quality caused mainly by pollution and the depletion of natural resources that could threaten the sustained production of the economy. Thus, a conventional SAM does not meet the economist need to model the environmental and natural resource implications of policy changes. To address this, Keuning (1992) proposed the National Accounting Matrix including Environmental Accounts (NAMEA) accounting framework. The NAMEA framework integrates accounts for pollutants and environmental impacts (in physical units) and economic accounts (in monetary units). That is, pollution impacts in physical units and economic flows in monetary units are combined into a single information framework. However, the NAMEA framework does not include accounts for pollution abatement activities and other environmental protection measures. Xie (2000) developed a framework for extending a SAM to include pollution-related information such as pollution abatement activities, sectoral payments for pollution abatement services, pollution emission taxes, pollution abatement subsidies, and environmental investment.

Xie's environmentally extended SAM (ESAM) distinguishes between conventional production activities and pollution abatement activities in the activity account. Specifically, the conventional commodity by pollution abatement activity matrix contains intermediate inputs of pollution abatement sectors. On the other hand, the pollution abatement commodity by production activity matrix reflects sectoral demand for/or spending on pollution abatement services. Thus, pollution abatement services are treated as special intermediate goods and added to the commodity accounts. The entry in the activity by pollution abatement commodity shows the total value of pollution abatement service or pollution clean-up. Household demand for pollution abatement services, if there is any, can be kept in the pollution abatement commodity by household matrix. The ESAM also has pollution tax accounts which receive payments from production sectors for their water pollution discharges. These taxes are transferred to the consolidated government accounts. Furthermore, the ESAM report the flow of pollutants and resources in physical units and keeps them outside the monetary SAM. This is different from the NAMEA approach where pollution data in physical units are mixed with monetary data in the SAM. In the ESAM, both production and consumption pollution are kept in the activity and consumption rows of the pollutants column. The pollutants row by activity column shows the level of pollution abatement. In the same manner, the depletion of natural resources as well as its renewal in physical units is presented in the resource row and column. It is important to note that the ESAM framework fails to represent the adverse impacts of pollution emissions on economic activities mainly due to data limitations. Thus, the framework serves as a sink for natural resource depletion and for pollution emission. Table 3.7 shows a representative ESAM framework.

Table 3.7: Framework of an Environmentally Extended SAM

Expenditures															
		Activity		Commodity		Factors		Institutions							
Receipts		Production	Abatement	Goods	Cleanup	Labour	Capital	Households	Government	Enterprise	ROW	S-I	Total	Pollutants	Resources
Activity	Production			Marketed output									Activity income	Production pollution	Resource consumption
	Abatement				Cleanup supply										
Commodity	Goods	Intermediate input (use)						Private consumption	Government consumption		Exports	Investment	Total demand		
	Cleanup	Pollution cleanup payment						Household payment for cleaning							
Factors	Labour	Factor payment											Factor income		
	Capital														
Institutions	Households					Factor income to households		Inter-household transfers	Transfers to households	Surplus to households			Household income	Consumption pollution	
	Government	Production tax & environmental tax		Tariff				Household income tax		Enterprise tax			Government income		
	Enterprise					Factor income to enterprises							Enterprise income		
	ROW			Imports		Factor income to ROW				Surplus to ROW			Foreign exchange flow		
S-I								Household savings	Government savings	Enterprise savings	Foreign savings		Savings		
Total		Activity		Supply expenditures		Factor expenditures		Household expenditures	Government expenditures	Enterprise expenditures	Foreign exchange flow	Investment			
Pollutants (in physical terms)		Pollutants abated or reused													
Resources (in physical terms)		Resource renewal													

3.7 **Constructing an Environmental SAM for the OWMA**

This study adapted the above described ESAM framework to construct an environmental SAM that captures the relationship between economic activities and water pollution and abatement measures for the case study area. The purpose is to provide a consistent database for calibrating the environmental CGE model and to quantitatively analyse the economic and environmental impacts of WQM policies at the basin level. The Olifants Environmental SAM (OESAM) includes three types of water pollutants (salinity, nitrogen, and phosphorus) and three corresponding pollution abatement sectors. These constitute the major pollutants in the case study area. The sources of economic data and estimation of environmental data used for the OESAM are discussed below.

Using the ESAM framework described in the preceding section, the OESAM is constructed by separating pollution abatement activities from economic activities. The SAM used is a consolidated version of three provincial SAMs[7] for the year 2012. The consolidated SAM has ten producing sectors (aggregated from the 46 sectors' provincial SAMs) with information on intermediate inputs, value added, consumption, taxes, and trade. The aggregation of sectors was done based on the type of pollutant discharged into the environment. The ten sectors are listed in Table 3.8. In addition to the ten producing sectors, the SAM is extended to include three pollutants, and three corresponding pollution abatement sectors.

Because a SAM with water pollution information for the OWMA has not been built before, we adopted an indirect approach to estimate the pollution-related data by pollutant and sector. Using data from the DWS Water Management System (WMS) for the year 2012, we estimated the load for each pollutant at selected monitoring sites along the Olifants River using the volume of flow and median concentration for the month. The estimated load from the monitoring sites was summed to obtain total load per pollutant for the base year. This was disaggregated by sector using the best available information from previous studies (DWS, 2011b, 2011c; Dabrowski & de Klerk, 2013). The pollution intensities of each pollutant by sector were then calculated using data on sectoral output and pollution load. The cost of operating a standard treatment plant to reduce pollutants to achieve in-stream water quality was obtained from previous DWS studies in the OWMA (DWS, 2003, 2011b). These data

[7] The initial SAMs were developed by the Development Bank of Southern Africa (DBSA) for the year 2006 (DWS, 2011b) and were updated by the author for the year 2012 using information from statsSA. The information includes data from supply and use tables, labour and household surveys, government accounts, and international trade accounts. The provincial versions of these sources were used to estimate the values for the case study area.

were disaggregated using the estimated pollution intensities to obtain the pollution abatement cost of each production sector and for each pollutant.

Table 3.8: Sector Classification

Number	10 Sectors	In 46-sector provincial SAM
1	Field crops	Cereal and Crop Farming; Sugarcane Farming
2	Horticulture crops	Citrus Farming; Sub-Tropical Farming; Vegetable Farming
3	Livestock	Livestock
4	Other agriculture	Game Farming; Forestry; Other Agriculture
5	Mining	Coal and Lignite Mining; Platinum Mining; Ferrous Mineral Mining; Non-Ferrous Mineral Mining; Other Mining and Quarrying
6	Chemical industry	Chemicals & Chemical Products (incl Plastic Products); Rubber Products, Fertilizers; Petroleum
7	Food, beverages and tobacco	Meat, Fish, Fruit, Vegetables, Oils and Fat Products; Dairy products; Grain Mill, Bakery and Animal Feed Products; Other Food Products; Beverages and Tobacco Products
8	Wood & paper	Wood and Wood Products; Furniture; Paper and Paper Products
9	Other manufacturing	Textiles, Clothing, Leather Products and Footwear; Publishing and Printing; Non-Metallic Mineral Products; Machinery & Equipment; Electrical Machinery & Apparatus; Communication, Medical and other Electronic Equipment; Manufacturing of Transport Equipment; Basic Metal Products; Structural Metal Products; Other Fabricated Metal Products; Other Manufacturing & Recycling
10	Services	Accommodation; Communication; Insurance; Real Estate; Business Services; General Government Services; Community, Social and Personal Services; Electricity; Water; Trade; Transport; Construction

As noted earlier, a SAM must be square with balanced corresponding rows and columns. That is, income must equal expenditure for each actor and supply must equal demand for each commodity and factor (Robinson et al., 1994). However, due to varying data sources, the initial OESAM did not fulfill the row-column constraint. Thus, we employed the cross-entropy method[8] (Robinson & El-said, 2000) to balance the micro-SAM using the GAMS software (Brooke et al., 1998). The environmental SAM constructed for the case study area is shown in Table 3.9.

[8] This method provides a flexible way to update and estimate a consistent SAM using information theory. Its advantage lies in the ability to incorporate both existing and new information about the various part of the SAM.

Table 3.9: An Environmental SAM for the OWMA (base year 2012, in Millions of Rands)

				Activities												
				Production sectors										Abatement sectors		
				1	2	3	4	5	6	7	8	9	10	11	12	13
Activities	Production	Field crops	1													
		Horticulture crops	2													
		Livestock	3													
		Other agriculture	4													
		Mining	5													
		Chemical industry	6													
		Food, beverages and tobacco	7													
		Wood & paper	8													
		Other manufacturing	9													
		Services	10													
	Abatement	Salinity	11													
		Nitrogen	12													
		Phosphorus	13													
Commodities	Goods	Field crops	14	43.3	1.3	1.3	2.2	5.9	3.2	375.8						
		Horticulture crops	15		262.8		29.5	28.4	8	201.7			77.3			
		Livestock	16			35.9	4.5	5.4	2.6	360.3		2.5	10			
		Other agriculture	17	24.1	108.3		152.6	17.1	3.7	2644.6	440.5	3.9	62.3			
		Mining	18	32.4	49.2	32.3	49.2	7849.2	485.2	2077.3	362	10171.3	3381.3			
		Chemical industry	19	715.1	852.5	771.9	557.6	200.4	1518.5	370.1	854.6	234.6	916.9			
		Food, beverages and tobacco	20		13.5	402.9	1148.1		79.7	3949	37.9	15.1	1902.6			
		Wood & paper	21	8.6	111.2	98.4	38.4	1234.3	34.8	297.4	1083.1	331	1342.8			
		Other manufacturing	22	692.8	1398.3	242.3	580.2	5067.6	250.5	473.5	209.5	10639.1	3106.8	272.4	187.6	145.9
		Services	23	466.8	1554.2	1017.4	1245.9	4454.5	645.3	5196.9	1263.7	6882.8	49574.7	480.1	264.1	295.2
	Cleanup	Salinity	24		109.4	96.9		1123.4	54.9	346.5	77.8	231				
		Nitrogen	25		85	79.8		790.4	64.2	653	75.9	530.6				
		Phosphorus	26		70.7	59.6		673.2	70	603.3	78.6	371.5				
Value Added	Labour	Highly skilled	27	94.5	217.9	194.1	224	7837.4	329.4	2438.7	299.4	3526.9	25402.5	186	264.2	215.3
		Skilled	28	172.4	196	123.3	263.8	6595.3	150.9	2736.3	416.5	2116.2	14388	169.6	240.8	196.1
		Unskilled	29	299.3	150.8	105.4	372.2	4651.7	50.7	3216.5	484	723.6	8118.3	134.9	191.6	156.2
		Capital	30	737.7	1204.1	690.5	1310.6	35487.5	546	8494.6	1216.9	12802.9	47641.7	803.5	1141.2	929.6
Institutions	Households	Poorest	31													
		Vulnerable	32													
		Middle income	33													
		High income	34													
		Enterprise	35													
		Government	36													
		ROW	37													
Taxes	Conventional taxes	Income tax	38													
		Activity tax	39	-22.9	-29	-22.8	-30.4	253.1	2.3	28.2	10.3	37.4	1136.4	-6.6	-10.6	-11.4
		Sales tax	40													
		Import tariffs	41													
	Pollution taxes	Salinity tax	42		27	24		276.8	13.5	85.3	19.2	56.8				
		Nitrogen tax	43		32	29.9		297.1	24.2	245.5	28.5	199.7				
		Phosphorus tax	44		119.2	100.4		1135.4	118.2	1017.7	132.7	625.4				
		S-I	45													
Total			46	**3264.1**	**6534**	**4084**	**5948**	**77984**	**4456**	**35812**	**7091**	**49502**	**157062**	**2040**	**2279**	**1927**

Table 3.9 continued

Commodities													Value Added			
Goods										Pollution abated			Highly skilled	Skilled	Unskilled	Capital
14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
3264.1																
	6534.4															
		4083.5														
			5948.4													
				77984.1												
					4455.8											
						35812.2										
							7091.1									
								49502.3								
									157061.6							
										2039.9						
											2278.9					
												1926.9				
													4701.4	12440	
												6742.7	3404.1	2552.2	
												5097.6	4311.3	3219.6	
												28534	14741.9		
															113006.8
1207.5	739	534.5	991.4	6028.8	8020.6	3493.4	5813.2	31810.3	37469.8				856	606.5	443.4	
105.7	42.6	195.6	155	123.8	284.1	366.8	387.1	468	607							
243.4	86.3	163.6	238.3	550.3	375.9	532.1	600.2	953	1547.5							
4821	**7402**	**4977**	**7333**	**84687**	**13136**	**40205**	**13892**	**82734**	**196686**	**2040**	**2279**	**1927**	**41230**	**27765**	**18655**	**113007**

Table 3.9 continued

Institutions							Taxes								Total
Poorest	Vulnerable	Middle income	High income	Enterprise	Government	ROW	Income tax	Activity tax	Sales tax	Import tariffs	Salinity tax	Nitrogen tax	Phosphorus tax	S-I	
31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46
															3264
															6534
															4084
															5948
															77984
															4456
															35812
															7091
															49502
															157062
															2040
															2279
															1927
2346.7	701.2	501.3	404.6			433.9									**4821**
559.1	299.1	145.3	1634.6			4156.5									**7402**
475	144.9	311.1	3393.7			231.3									**4977**
1581.7	866.1	546.6	670			211.6									**7333**
						53732.8								6464.8	**84687**
1799.1	2039.6	1267.7	780.5			257.3									**13136**
6854	5521.3	4588.9	12734.1			2957.4									**40205**
1010.2	1538.7	1119.4	3140			1144.6								1358.7	**13892**
3053.3	2725.1	4298.2	10505.2		5139	15522.9								18223.4	**82734**
3818.6	6150.8	7231.2	31265.4		41701.9	23317.6								9858.8	**196686**
															2040
															2279
															1927
															41230
															27765
															18655
															113007
			4154.8	4281.7	416.1										**25994**
		1224		11324.2	289.7										**25537**
	941.2			12142.3	275.8										**25988**
747.1				45867.3	173										**90063**
															113007
							35475.3	1334	2735.7	5290.6	502.6	856.9	3249		**49444**
				13009.9	2801.4										**113826**
2471.9	2618.7	2731.7	11385.1	16267.9											**35475**
															1334
															2736
															5291
															503
															857
															3249
1277.3	1990.2	2022.4	9995.3	10113.5	-1352.8	11859.8									**35906**
25994	**25537**	**25988**	**90063**	**113007**	**49444**	**113826**	**35475**	**1334**	**2736**	**5291**	**503**	**857**	**3249**	**35906**	**1549967**

3.8 **Calibration of the Olifants Environmental CGE Model**

The regional environmental CGE model presented in sections 3.2 and 3.3 contains a number of parameters. Correct identification of these parameters is important because the parameter values influence the results of policy simulations by determining the reaction of economic agents to the price changes induced by the environmental policy. This section provides a brief introduction to the two alternative approaches used to numerically specify CGE models. The review provides motivation for the approach taken to calibrate the Olifants environmental CGE model.

The econometric and calibration approaches are the two methods used in the literature to parameterize CGE models[9]. In the econometric approach, parameter values are estimated by statistical methods using more than one-year time series data. From a statistical perspective, this is satisfactory because the modeller can implement statistical tests of the parameters to provide standard errors and confidence intervals (Lau, 1984). Moreover, the approach enhances the empirical relevance of CGE models by allowing for a more flexible specification of production and consumption functions (Bergman, 1990). Nonetheless, the econometric approach is not commonly used in the literature because it is sophisticated and requires long time series data. The calibration approach, on the other hand, is widely used in the CGE literature because it is simple and less demanding of data. It involves identifying the model parameters using one-year information contained in a SAM with the assumption that the economy under consideration is in equilibrium. Thus, parameter values are specified in such a way that the model replicates the economy represented in the SAM. However, due to the limited number of data contained in the SAM, some parameter values (usually elasticities) are assigned extraneously. That is, the calibration approach suffers from the parameter under-identification problem (Lau, 1984). Furthermore, there is no statistical test of model specification and the approach requires restrictive assumptions about technology and preferences. Despite these weaknesses, the calibration approach is employed in nearly all CGE applications (Robinson et al., 1999) and is adopted to parameterize the environmental CGE model developed here.

The Olifants environmental CGE model is calibrated to the OESAM by first defining all prices to be equal to one as commonly done in the calibration approach. This implies that all price indices are indexed to the model's base year and that sectoral flows in the OESAM

[9] See Mansur and Whalley (1981) for a thorough comparison of the two approaches.

measure both real and nominal magnitudes (Robinson et al., 1999). Factor returns were estimated based on the procedure described by Robinson et al. (1999) with factor quantities sourced from Hassan and Thurlow (2011). The model has two different types of parameters. The first which consist of share parameters like consumption shares and average tax and savings rates were determined from the data contained in the OESAM. The other set of parameter values comprises output and trade elasticities and were adopted from Hassan and Thurlow (2011). Table 3.10 summarizes the range of elasticities used in this study.

Table 3.10: Values for Key Elasticity in the Model

Elasticity	Values
CES between factor inputs	0.7
CES between imported and domestic commodities	1 to 4
Export demand elasticity	1 to 4

The estimated parameters may contain uncertainty which may affect the results of policy simulations. Thus, sensitivity analyses were conducted to check the reliability and robustness of the simulation results to changes in model parameters and assumptions.

3.9 **Summary**

This chapter presented the technical specifications of the environmental CGE and SAM models for the Olifants basin. The model integrates water pollution related information such as pollutants, pollution taxes, and abatement activities of polluting sectors into a standard CGE model for the basin. The environmental SAM serves as a consistent database for calibrating the regional model and makes it possible to quantitatively assess the trade-offs (social, economic, and environmental) associated with the implementation of WQM policies in the basin.

CHAPTER 4: THE ECONOMIC IMPACT OF PROTECTING WATER RESOURCES IN THE OLIFANTS RIVER BASIN

4.1 Introduction

This chapter employs the Olifants environmental CGE model as described in the previous chapter to analyse the trade-offs between economic growth and the protection of environmental quality in the basin. Specifically, the chapter will focus on the economic impact of taxing water pollution through the interactions between water pollution control, economic growth, sectoral structure, employment, and international trade. Thus, the analysis in this chapter is used to answer research questions 1 and 2 as formulated in chapter 1.

Although both regulatory and market-based instruments are used to manage water quality in SA, market-based instruments have been shown to be more cost-effective and promising instruments for achieving sustainable water quality management (Baumol, 1977; DWS, 2016). Thus, in this chapter, we analyse the economic and environmental consequences of taxing water pollution in the Olifants river basin, the third most water-stressed and polluted basin in SA. In addition, we compare two revenue recycling scenarios to ameliorate the adverse impact of the pollution tax on the regional economy. It is important to state that the use of the tax instrument in this study is not an endorsement of this particular policy instrument but rather a convenient way to represent the suite of market-based instruments that provide economic incentive to modify behaviour, including tradable emission permits.

Section 4.2 describes the policy scenarios and model closures adopted for the analysis. The results of the model simulations are presented and discussed in section 4.3. In section 4.4, a range of sensitivity analyses is performed to test the robustness of the results to changes in parameter values and assumptions. The chapter concludes in section 4.5 with a summary.

4.2 Policy Scenarios

Three policy regimes for reducing nutrient load (total nitrogen and total phosphorus) in the study region are implemented and compared. In all three scenarios, it is assumed arbitrarily that the government raises the pollution tax rate on nutrient load by 50% with reference to the base value. The first scenario assumed that all revenue generated from the pollution tax is absorbed in the government budget balance. Results of this scenario are reported under the "*no-revenue recycling*" scenario. Two other policy scenarios are tested to evaluate two alternative options for recycling tax revenue to mitigate the impact of the pollution tax on economic activity. One of the complementary policies tested is to reinject the pollution tax proceeds back into the economic system by recycling all revenue through a direct subsidy to consumers as a lump-sum *transfer to households*. The second tax revenue recycling regime is to return the tax revenue to pollution abatement sectors in the form of *production subsidy*. Thus, there are three policy simulations including the reference scenario where the government does not recycle the tax revenue (i.e. the pollution tax revenue is used for fiscal adjustment). It is important to state that the different policy scenarios represent alternative closure rules for the macroeconomic equilibrium conditions in the model. For the factor market equilibrium constraint, we assumed that higher skilled labour (highly skilled and skilled) and capital are fully employed with flexible real wages and capital rental price. On the contrary, and to reflect the reality in the SA labour market, unskilled labour is assumed to be not in full employment at a fixed real wage. The Consumer Price Index (CPI) is used as the numeraire.

Due to space limitations and the fact that the results follow a similar pattern, we report and discuss here only results of simulations of the pollution tax policy on nitrogen. The simulation results on phosphorus and salinity are presented in the appendix. The proposed tax rate on nitrogen emissions used amounted to R 0.6 per kilogram. While using a single test pollutant is valid for illustration, it is acknowledged that it would be interesting to investigate the combined effects of a simultaneous imposition of the tax on multiple pollutants.

4.3 Results and Discussion

The Olifants environmental SAM (OESAM) which is a statistical representation of the economic and social structure of the Olifants river basin contains certain structural and economic relationships, which will partly drive how the basin's environmental CGE model responds to the policy simulations. Thus, this section begins with a description of the economic structure and pollution-related information embodied in the OESAM to help explain the outcomes of the policy simulations.

Table 4.1 presents the economic structure and pollution-related information in the base year. The total value of economic activities was approximately R215 billion which represent about 7% of the national GDP in 2012. It comprises of households' spending R126 billion (58.6%), government spending R47 billion (21.9%), investment spending R36 billion (16.7%), and net exports R6 billion (2.8%). Economic activity in the basin is driven by the services sector which includes general government services (46.1%), followed by the mining sector (26.6%) and the manufacturing sector (21.5%). Of the four manufacturing sectors, food, beverages, and tobacco and other manufacturing contribute the most (9% and 9.9% respectively). Agriculture share in RGDP is 3.7% with horticultural crops and other agriculture respectively contributing 1% and 1.2%. The total wage bill (i.e. employees' compensation) was R88 billion comprising of highly skilled labour R41 billion (47%), skilled labour R28 billion (31.7%), and unskilled labour R19 billion (21.3%). The services sector had the largest share of the wage bill (55%) followed by the mining sector (22%). The total value of capital was R113 billion making the capital to labour ratio almost 55:45. The share of the three pollution abatement activities in RGDP is 2.2%. Table 4.1 also shows that manufacturing (primarily food processing and other manufacturing activities), agriculture (especially horticulture and livestock sectors), and mining are the sources of water pollution in the study area. Together, these sectors generate over 90% of the pollution in the basin. Furthermore, polluting sectors are relatively capital intensive and contribute marginally to the employment of unskilled labour.

The OESAM also provides information about each sector's trade orientation. The major foreign exchange earners in the basin (i.e. in terms of share of output exported) are horticultural crops (63.5%), mining (68.9%), and other manufacturing (31.9%) (See column 10 of Table 4.1). On the import side, the major importing sectors (i.e. in terms of share of import in domestic demand) include chemical manufacturing (66.8%), other manufacturing (49.2%), and wood and paper (52.0%) (See column 12 of Table 4.1). The trade information reveals that polluting sectors generate the bulk of the basin's export earnings and are major users of imported commodities thus; they will be strongly impacted by an increase in the pollution tax relative to trade exposed non-polluting sectors. The reason is that the responsiveness of trade ratios to an increase in pollution tax depends on trade shares and elasticities. Thus, for a given trade elasticity, an increase in the pollution tax will change the ratio of domestic to external prices making domestic commodities to be more expensive. These structural features are the key drivers of the economic impacts of taxing water pollution in the basin.

4.3.1 Microeconomic Impact

Table 4.2 shows that the pollution tax achieves its environmental objective of reducing water pollution in the Olifants River. In each scenario, the total nitrogen discharged is reduced. The environmental goal of protecting water quality, however, comes at an economic cost of a lower real regional gross domestic product (RRGDP), as the imposition of the tax raises the cost of production in polluting sectors causing them to re-optimize at lower output levels. The magnitude of the cost of production depends among other things on pollution intensities and abatement costs. Thus, polluting activities with higher pollution intensity and abatement cost are severely impacted. On the other hand, non-polluting sectors (such as field crops and services) record an increase in production due to the fall in their production costs, compared with polluting activities. The fact that the “other agriculture sector” buys over 40% of its intermediate inputs from the chemical and other manufacturing sectors (polluting upstream sectors) explains the drop in production of this activity, highlighting the indirect impact of the tax through increased production costs in upstream input supply sectors.

In the new equilibrium, demand for pollution-intensive goods (such as horticultural crops and chemical manufacturing) declines (both domestically and export) due to increasing domestic prices. This leads to a fall in domestic production of polluting activities, negatively impacting demand for primary factors, particularly capital[10], and in turn, reducing remuneration to factors of production and household income. In general, the water pollution tax leads to an increase in the relative price of exports which prompts a depreciation of the real exchange rate (i.e. makes imports more attractive). This confounds the impact on polluting sectors as they are the most trade-exposed, i.e. high export and import components.

[10] Though prices of both capital and labour drop, capital bears the burden of the tax increase, because the reduction in capital by polluting firms outweigh the reduction in labour since polluting firms are relatively capital intensive (see Table 4.1). This is reflected in a fall in the economy-wide capital to labour price ratio.

Table 4.1: Economic Structure and Pollution-Related Information in the Study Area (base year 2012)

Sectors of economic activity	%age shares in RGDP	%age shares in total emissions		Emission intensity (kg/Rands)		Factors' shares in industry costs (%)		%age shares in total exports	%age share of exports in industry	%age shares in total imports	%age shares of imports in domestic demand
		Nitrogen	Phosphorus	Nitrogen	Phosphorus	Unskilled labour	Capital				
Agriculture	**3.7**	**26.58**	**26.18**	**0.0243**	**0.0213**	**4.70**	**19.83**	**4.9**	**20.50**	**3.7**	**14.17**
Field crops	0.8	0	0	0	0	9.2	22.5	0.4	13.3	1.3	33.9
Horticultural crops	1.0	13.63	13.58	0.0233	0.0193	2.3	18.4	4.1	63.6	0.8	25.8
Livestock	0.7	12.95	12.60	0.0350	0.0260	2.6	16.9	0.2	5.7	0.6	15.4
Other agriculture	1.2	0	0	0	0	6.3	22	0.2	3.5	1.0	17.7
Mining	**26.6**	**17.68**	**18.06**	**0.0182**	**0.0154**	**6.0**	**45.4**	**52.6**	**68.9**	**6.3**	**21.4**
Manufacturing	**21.5**	**55.74**	**55.76**	**0.0298**	**0.0259**	**4.65**	**23.73**	**19.7**	**13.4**	**50.8**	**32.70**
Chemical manufacturing	0.9	3.15	4.08	0.0258	0.0281	1.1	12.2	0.3	5.7	8.4	66.8
Food, beverage and tobacco	9.0	26.15	28.64	0.0327	0.0301	9.0	23.7	2.9	8.2	3.6	11.0
Wood and paper	1.7	2.80	3.44	0.0192	0.0198	6.9	17.1	1.1	16.0	6.1	52.0
Other manufacturing	9.9	23.64	19.60	0.0192	0.0133	1.5	25.7	15.4	31.9	32.7	49.2
Services	**46.1**	**0**	**0**	**0**	**0**	**5.2**	**30.3**	**22.8**	**14.8**	**39.2**	**22.6**

Source: Olifants environmental SAM, 2012; Notes: Sectors with zero emissions implies negligible share in total emissions

The horticultural crop and chemical manufacturing sectors are severely affected due to their high trade shares (see Table 4.1). Mining and horticultural crop production for instance, which export over 60% of their output became less competitive and lost market shares on the international market. Similarly, domestic production was readily replaced by cheaper imports in manufacturing, particularly the chemical and wood and paper sectors, which source very high shares of their inputs' demand from imports (see Table 4.1).

Table 4.2: Micro Level Impacts of a 50% Increase in Pollution Tax on Nitrogen Emission under Alternative Revenue Recycling Scenarios (%age Change Relative to Base-run)

	Base-run (Units)[a]	Scenario 1 No-revenue recycling (%age change)	Scenario 2 Uniform transfers to households (%age change)	Scenario 3 Production subsidy to pollution abatement sectors (%age change)
Total nitrogen discharged	1.805	-0.33	-0.26	-0.21
Changes in sectoral output				
Field crops	3.273	0.17	0.47	0.05
Horticultural crops	6.557	-1.57	-1.49	-0.33
Livestock	4.093	-0.37	-0.36	-0.10
Other agriculture	5.975	-0.11	0.08	-0.04
Mining	78.186	-0.06	-0.03	0.03
Chemical manufacturing	4.478	-1.04	-0.75	-0.18
Food, beverage and tobacco	35.997	-0.33	-0.22	-0.12
Wood and paper	7.124	-0.45	-0.32	-0.04
Other manufacturing	49.378	-0.42	-0.38	-0.27
Services	157.614	0.21	0.24	0.06

Source: Olifants environmental CGE model

a. Except for total nitrogen discharged (in 1000 kilograms), base year units are in millions of Rands.

Comparing across scenarios, recycling the tax revenue leads to favourable economic outcomes relative to the no-revenue recycling case. However, the no-revenue recycling scenario achieves a larger environmental dividend highlighting the trade-offs between economic growth and environmental protection objectives. Under the no-recycling scenario, the model solves by clearing the government account leading to higher government balance (savings) as government consumption is fixed and transfers are held constant (equation 32 in chapter 3). Higher government savings, in turn, will lead to lower savings by households to maintain the economy-wide saving-investment balance, since total investment levels are kept exogenously fixed (equation 33 in chapter 3). Lower private savings' levels leave households with more disposable income to spend on household consumption (equation 24 in chapter 3). This represents an indirect subsidy to consumer demand, which mitigates the negative impacts of the tax on economic activity.

Under scenario 2, the revenue from the water pollution tax is returned to the economy as uniform government transfers to households. Compared with the no-revenue recycling scenario which can be considered an indirect subsidy to households in the form of an income tax break, this scenario is a direct subsidy to households in the form of cash grants. Households' income is boosted by the transfers which increase their purchasing power and enhance demand for consumption goods. As a result, production is stimulated in both polluting and non-polluting sectors positively impacting real regional GDP.

In the third scenario, the pollution tax revenue is returned to pollution abatement sectors in the form of production subsidy. This is a supply-side subsidy that reduces the cost of production in pollution abatement sectors, thus boosting the capacity of the regional economy to clean up. The effect is a marginal increase in production in polluting sectors softening the negative impact of the tax policy. As a result, the output of polluting sectors falls by a smaller margin compared with the previous two scenarios. Since this study assumes that the output of pollution abatement sectors is a special intermediate good bought by polluters, the fall in their prices due to the increased government subsidy encouraged polluters to increase their demand.

4.3.2 Macroeconomic Impact

Table 4.3 displays results for the macro level impact. The results follow the same trend as in the sector level analysis. Without revenue recycling, the regional economy contracts but the magnitude of reduction in the RRGDP is modest. The contraction is due to the fact that the negative impact of the environmental policy on polluting sectors outweighs the gains by non-polluting sectors. This result, therefore, suggests that implementing pollution control policies in the basin without complementary measures to address the potential negative impact would lead to the contraction of the regional economy. Nonetheless, recycling the tax revenue would mitigate the adverse economic impact of the environmental policy with a trade-off between improving the current welfare of the basin's population and boosting the capacity of the regional economy to clean up its pollution.

It is important to state that the macro closure choice we made drive some of these results. The choice is influenced by our understanding of how the regional economy operates. Under scenario 1, we assume that the government is more concerned about fiscal adjustments such as reducing the budget deficit. As shown by the government savings row, the government deficit is reduced by 29%. In the second scenario, we assume that the government is not only concerned about protecting water quality but also interested in the economic welfare of the population. As a result, the tax revenue is transferred to households in order to achieve fiscal neutrality. In essence, household consumption expenditure is boosted causing the regional economy to expand to offset the negative impact of the pollution tax. The third scenario assumes that the government is interested in supporting producers of pollution abatement goods so as to boost the capacity of the regional economy to clean up. This reduces the cost of pollution abatement goods and as a result, the economic cost of the pollution tax is softened.

Table 4.3: Macro Level Impacts of a 50% Increase in Pollution Tax on Nitrogen Emission under Alternative Revenue Recycling Scenarios (%age Change Relative to Base-run)

	Base-run (Units)[a]	Scenario 1 No-revenue recycling (%age change)	Scenario 2 Uniform transfers to households (%age change)	Scenario 3 Production subsidy to pollution abatement sectors (%age change)
RRGDP	214.947	-0.03	0.02	-0.01
Private consumption	125.623	-0.05	0.04	-0.013
Exports	102.360	-0.10	-0.06	-0.02
Imports	96.357	-0.11	-0.06	-0.02
Real exchange rate	1.0	0.012	0.021	0.001
Capital rental price	1.0	-0.32	-0.22	-0.098
Wage rate highly skilled labour	1.0	-0.21	-0.15	-0.070
Wage rate skilled labour	1.0	-0.26	-0.19	-0.081
Unskilled labour employment	16.805	-0.21	-0.34	-0.06
Government revenue	46.876	0.66	0.79	0.86
Aggregate government transfer	1.147	0	39.33	0
Government savings (surplus)	-1.354	-28.64	0	-8.82
Aggregate household savings[b]	25.689	-1.31	0.10	-0.18
Total absorption	208.944	-0.03	0.02	-0.01

Source: Olifants environmental CGE model

a. Except for employment (in thousands of workers), base year units are in millions of Rands.
b. Aggregate household savings include savings by enterprises.

As mentioned before, our factor market closure assumes a perfectly elastic supply of unskilled labour with a fixed real wage for unskilled workers. However, the supply of higher skilled labour (highly skilled and skilled) is assumed to be inelastic with flexible real wages.

The economic contraction causes the wage rates of higher skilled labour to fall with the return to skilled labour declining more. This is because polluting sectors employ a greater share of skilled labour relative to the non-polluting sector so their contraction as a result of the environmental policy led to excess supply in the economy. Demand for unskilled labour also falls for the same reason causing unskilled unemployment. Although the economy expands under scenario 2, the demand for unskilled labour falls by a greater percentage relative to the other scenarios. The reason is that the government transfer to households stimulates demand for imported commodities at the expense of domestic production mainly due to higher domestic prices and the fact that polluting sectors are more trade-exposed. This result implicitly assumes that foreign countries (i.e. other regions in SA and the rest of the world) have lower or no taxes applied to their pollution-intensive commodities. In such a case, recycling the tax revenue through lump-sum transfers would be a poor policy option if the government is concerned about job losses as a result of the tax policy.

The environmental policy also has an impact on the basin's trade patterns through its impact on the external balance which consists of the trade balance and transfers between institutions in the basin and the rest of the world (which includes the rest of SA and other countries). The pollution tax raises the cost of domestic production making exports expensive and causing disequilibrium between exports and imports. We assume that the external balance is exogenous thus; the exchange rate must depreciate to maintain balance in the current account. This explains the depreciation of the real exchange rate under the three scenarios.

It is also important to note that the analysis in this chapter underestimates economic gains from improved water quality, as we do not account for beneficial feedback effects of reduced water pollution, such as positive impacts on human health and aquatic ecosystems. Previous attempts to estimate such benefits in SA indicate that they are significant, and may exceed economic costs of the water pollution tax policy, estimated above. Employing the cost of illness approach (COI),[11] De Lange *et al.* (2012) estimated the direct and indirect costs of microbial pollution in the Olifants river basin to amount to, respectively, R0.704 and R1.141 billion per year. In another study, the department of water and sanitation (DWS, 2003) estimated the downstream benefits of reducing salinity in SA to be R0.467 billion per year. Another caveat to note in considering the results of this study, however, is the fact that our

[11] The COI approach measures benefits of pollution prevention by estimating the direct and indirect costs of associated illnesses avoided. Direct costs are typically estimated using direct healthcare expenditures on treating waterborne diseases, such as cholera and diarrhoea. Indirect costs are estimated based on a composite measure of the burden of the disease, e.g., forgone income due to illness (De Lange *et al*., 2012).

analysis has not covered a major water pollution source in the Olifants, which is municipal waste.

4.4 **Sensitivity Analysis**

In this section, we examine the robustness of the main model results for different parameter values and assumptions about the factor market. The parameters of concern are the elasticities of factor substitution and trade, and pollution intensity. As mentioned earlier, these parameter values are based on estimates with some level of uncertainty, which may affect the results of policy simulations. Therefore, we investigate in this section implication of two alternative values for each parameter: a high value where the base value is increased by 20% and a low value where it is reduced by the same magnitude. For simplicity, the analysis is only undertaken for the no-revenue recycling scenario.

The elasticity of factor substitution describes the ease with which producers in the model can substitute among the different factors of production in the second level of the technology nest to produce aggregate value-added, i.e., substitution among capital and highly skilled, skilled, and unskilled labours. Put differently, it determines the flexibility of the production technology. A high value implies a flexible production technology where it is easy to substitute among factors whereas; a low value indicates a relatively rigid technology. The results for this sensitivity analysis are presented in Table 4.4. It shows that changing the value of the elasticity of factor substitute parameter influences the proportion of the burden of the pollution tax born by the factor intensively used in the polluting sectors. Under a flexible production technology, capital (which is intensively used in polluting sectors) bears relatively less burden of the pollution tax because polluting sectors can easily substitute among factors so their contraction affects all factors proportionately. This explains the bigger fall in the returns to higher skilled labour and unskilled labour employment under the high-value scenario. On the other hand, it is difficult to substitute among factors when the production technology is rigid so capital bears a greater burden of the pollution tax explaining why the rental price of capital fell by a bigger percentage under the low-value scenario. The result also shows that this parameter value influences the magnitude of the economic cost (i.e. loss in RRGDP) of the pollution tax. The economic cost is lower in the rigid scenario compared with the flexible scenario. The reason is that non-polluting firms can comparatively increase their demand for capital because it is cheap but it requires a complementary increase in the demand for higher skilled labour to operate the additional capital. This explains why the returns to higher skilled labour rose under the rigid scenario relative to the flexible and reference scenarios. Nonetheless, the trend of changes in key model results under both high and low-value scenarios are consistent with the reference scenario.

Table 4.4: Results of the Sensitivity Analysis of the Elasticity of Factor Substitution

	Reference scenario	High value scenario	Low value scenario
RRGDP	-0.03	-0.06	-0.01
Capital rental price	-0.32	-0.29	-0.45
Wage rate highly skilled labour	-0.21	-0.26	0.12
Wage rate skilled labour	-0.26	-0.28	-0.19
Unskilled labour employment	-0.21	-0.58	-0.05

Source: Olifants environmental CGE model

We also tested for the potential impacts of changing trade elasticities on the model results. The results for the sensitivity analysis of changes in trade elasticities are presented in Table 4.5. The results show that high trade elasticities yield lower real regional GDP for the reason that it becomes easier for consumers to substitute towards imported commodities when domestic prices rise due to the introduction of the pollution tax. This result indicates that high trade elasticities would cause production in polluting sectors to move beyond the basin's borders with the consequent negative impact on employment. Thus, implementation of a pollution tax in the presence of high trade elasticities without revenue recycling will increase the economic cost of reducing water pollution.

Table 4.5: Results of the Sensitivity Analysis of Trade Elasticities

	Reference scenario	High value	Low value
RRGDP	-0.03	-0.04	-0.02
Import	-0.11	-0.17	-0.08
Export	-0.10	-0.16	-0.07
Exchange rate	0.011	0.008	0.013

Source: Olifants environmental CGE model

Next, we checked the sensitivity of the model results to variations in the pollution intensity coefficients. These parameters measure the amount of pollutant g generated by producing a unit of commodity c by activity a. In this study, the pollution intensity coefficient of a given pollutant is assumed to be linearly related to activity level (output) thus, a high value increases total emissions and vice versa. The result of this sensitivity analysis is therefore straightforward: the economic cost of the water pollution tax increases with increasing pollution intensity and vice versa (4.6). The result also highlights that a higher pollution tax will be required to reduce the total pollution discharged by polluters.

Table 4.6: Results of the Sensitivity Analysis to Changes in Pollution Intensity

	Reference scenario	High value	Low value
RRGDP	-0.03	-0.05	-0.01
Unskilled labour employment	-0.21	-0.43	-0.04
Total nitrogen discharged	-0.33	-0.66	-0.07

Source: Olifants environmental CGE model

The assumption of perfectly mobile capital in our reference scenario might only apply in the medium to long run but in the short run, capital is assumed to be fixed. Given that our model is static, we test the effect of a non-mobile sector-specific capital assumption. That is, all labour types are perfectly mobile but capital is sector-specific. Results of this sensitivity analysis indicate a lower economic cost of the pollution tax (4.7). The reason is that each activity in the model is forced to employ its observed base year capital thus; capital bears a lesser burden of the tax.

Table 4.7: Results of the Sensitivity Analysis of a Change in Capital Mobility Assumption

	Reference scenario	Sector-specific capital scenario
RRGDP	-0.03	-0.002
Wage rate highly skilled labour	-0.21	-0.17
Wage rate skilled labour	-0.26	-0.22
Wage rate unskilled labour	0	-0.27
Total nitrogen discharged	-0.33	-0.10

Source: Olifants environmental CGE model

4.5 Summary

The purpose of this chapter was to use the Olifants environmental CGE model developed in chapter three to evaluate the economic and environmental implications of a tax on water pollution policy. The simulation results show that internalising the negative externality of water pollution in the Olifants river basin will effectively reduce pollution discharge (i.e., achieve its environmental goals). As expected, the pollution tax policy changes the structure of economic incentives in favour of less polluting sectors. Environmental protection, however, is achieved at some cost (not accounting for the potential benefits of clean water) to the regional economy, e.g., loss in RGDP. The economic burden of the pollution tax happens to be insignificant though, due to the small relative share of the water pollution supply and abatement costs in total production costs.

Simulations of alternative programs for recycling the revenue from the pollution tax suggest a high potential for fiscal policy regimes to mitigate the economic burden of the tax. The negative impact of the pollution tax on economic activity was totally offset by a tax revenue recycling regime of direct transfer to consumers. This fiscal programme also achieved relatively bigger reductions in the emission of pollutants, compared with the subsidy to abatement activities revenue recycling option.

Results of the sensitivity analyses show that the model outcomes are stable across a reasonable range of parameter values and assumptions. That is, changes in parameter values and assumptions only influence the magnitude of the impacts but not the direction (at least for the scale of values in which our simulation analyses occurs).

CHAPTER 5: THE DISTRIBUTIONAL IMPACTS OF TAXING WATER POLLUTION IN THE OLIFANTS RIVER BASIN

5.1 Introduction

The distributional burden associated with water quality management interventions in the Olifants river basin will vary across households or socioeconomic groups for two reasons. Firstly, their introduction will change factor prices (especially in a general equilibrium setting) and households with greater income shares derived from factors whose returns fall will suffer the most. Secondly, households differ in expenditure patterns in terms of the proportion of income spent on pollution-intensive commodities. Thus, for a proper assessment of the welfare impacts of these measures, both the income and expenditure channels must simultaneously be considered. Moreover, gaining deeper insights into the distributional impacts of these interventions is of great policy relevance considering that their social acceptance is highly dependent on their perceived impact on the poor and vulnerable groups in society. In this context, this chapter applies the Olifants environmental CGE model to assess the distributional impacts of introducing a water pollution tax to protect aquatic ecosystems in the basin by considering both the income and spending-side effects. In so doing we address research question 3 as formulated in chapter 1.

Both partial and economy-wide modelling approaches have been employed to study the distributional impacts of environmental policies. Partial equilibrium models have been, however, considered deficient as they do not account for economy-wide implications and also they only consider the distributional effects resulting from a change in households' expenditure on taxed commodities (Beck et al., 2015). Economy-wide models such as input-output and computable general equilibrium (CGE) models, on the other hand, consider how households earn and spend their incomes as well as indirect effects and are widely employed to trace the distributional impacts of environmental policy interventions. In the CGE approach, two methods exist for analysing the impacts of policy interventions on the distribution of incomes across households. The first is the traditional method where the household sector is disaggregated into a number of representative households normally based on considerations like income and characteristics of the household head. The second is the micro-simulation method where the CGE model is integrated with a household module in which the units correspond to individual household observations in a nationally representative survey (Löfgren, 2003). The household module may be fully integrated with the CGE model or sequentially linked to it. In this chapter, the representative household method is employed with the Hicksian equivalent variation[12] as a welfare indicator.

The existing literature on the distributional impacts of environmental taxes particularly carbon and energy taxes, suggest they are generally regressive i.e. it hurts the poor more than the rich (see e.g., Poterba, 1991; Pearson & Smith, 1991; Hamilton & Cameron, 1994; Baranzini et al., 2000; Dinan & Rogers, 2002; Brannlund & Nordstrom, 2004; Wier et al., 2005; Kerkhof et al., 2008; Ojha, 2009; Devarajan et al., 2011; Rausch et al., 2011; Wang et al., 2016). Studies on the distributional impacts of other environmental taxes such as water pollution taxes are scarce. Thus, the analysis in this chapter contributes to the literature and also investigates whether similar conclusion could be drawn for taxing water pollution in the Olifants river basin.

Section 5.2 describes the policy scenarios and the disaggregation of the household sector to investigate the distributional impacts. Section 5.3 reports and discuss the results of the model simulations while section 5.4 contains a summary of the chapter.

5.2 **Policy Scenarios**

Three policy scenarios are run to assess the distributional impacts of introducing a water pollution tax to reduce nutrient load (total nitrogen and total phosphorus) in the study region. In all three scenarios, it is assumed arbitrarily that the government raises the pollution tax rate on nutrient load by 50% with reference to the base value. The first scenario assumed that all revenue generated from the pollution tax is absorbed in the government budget balance. This was intended to reveal the absolute incidence[13] of the tax or the direction of the distributional burden when the tax revenue is not returned to the economy. Results of this experiment are reported under the "*no-revenue recycling*" scenario. Implications of two other policy scenarios for the differential incidence of the pollution tax have been analysed to compare alternative remedial options for recycling the tax revenue[14]. One of the complementary policies tested was to reinject the pollution tax proceeds back into the economic system by recycling all revenue through a direct subsidy to consumers as a lump-sum *transfer to*

[12] Equivalent variation measures the change in expenditure at base year prices that would be equivalent to the policy implied change in utility. Put differently, it measures the monetary value of a change in utility for a given household as a result of introducing the water pollution tax.

[13] Fullerton and Metcalf (2002) distinguished three tax incidences in terms of distributional effects: absolute, balanced budget and differential.

[14] Revenue recycling is a prominent issue analysed in the literature because it has important implications on the equity of environmental policies (see e.g., Goulder, 1998; Bovenberg & Goulder, 2002; Baranzini et al., 2000; Oladosu & Rose, 2007; Ojha, 2009; Beck et al., 2015; Yusuf & Resosudarmo, 2015).

households. The second tax revenue recycling regime was to return the tax revenue to pollution abatement sectors in the form of *production subsidy*.

For the purposes of investigating distributional impacts, households are disaggregated into four income groups representative of those living in the Olifants river basin, using monthly poverty lines (lower and upper-bounds) in 2018 prices from Statistics South Africa (StatsSA, 2018). Households with income levels below the lower-bound poverty line (R 785 per month) have been grouped into the *poorest* category. A *vulnerable* households' category has been defined to include those with incomes above the lower-bound poverty line but far below the upper-bound poverty line (i.e. between R 785 – R 825 per month). Note that these two groups (poorest and vulnerable) constitute more than 75% of the population in the study area (see Table 5.1). *Middle-income* households are those with incomes far above the lower-bound poverty line (i.e. above R 825 – R 1183 per month) whereas *High-income* households are those above the upper-bound poverty line (R 1183 per month).

5.3 **Results and Discussion**

This section begins with a description of the different household groups in terms of their income sources and spending patterns. As Table 5.1 shows, sources of income in the basin vary significantly between the above-defined income groups. *High-income* households derive a significant share of their income from highly skilled labour and capital, whereas the *poorest* households derive the bulk of their income from unskilled labour and transfers (both government and inter-institutional). On the consumption expenditure side, Table 5.2 indicates that the share of pollution-intensive goods in the consumption basket of *poorest* households is greater than that of *high-income* households. As will become clear in the subsequent policy simulation analysis, these structural features (household sources of income and share of pollution-intensive goods in household consumption basket) are key determinants of the distributional impacts (burden) of taxing water pollution in the river basin.

Table 5.1: Household Income Source and their Tax Shares (base year 2012)

Household	Household income (upper)	Labour income share				Capital income share	Transfer income share	Tax share of income	Population share
		Highly skilled	Skilled	Unskilled	Total labour income share				
Poorest	10,600	0.00	0.11	0.55	0.66	0.17	0.18	0.085	0.26
Vulnerable	18,400	0.19	0.13	0.17	0.50	0.44	0.06	0.103	0.51
Middle income	38,000	0.24	0.17	0.08	0.49	0.47	0.05	0.105	0.12
High income	Above 38,000	0.28	0.10	0.00	0.38	0.61	0.01	0.126	0.09

Source: Olifants environmental SAM, 2012 and DWS, 2011b; Notes: Household income is in Rands per annum.

Table 5.2: Household Consumption Shares (base year 2012)

Household	Field crops	Horticultural crops	Livestock	Other agriculture	Chemical manufacturing	Food, beverage and tobacco	Wood and paper	Other manufacturing	Services	Share of pollution intensive goods in household consumption basket
Poorest	0.11	0.03	0.02	0.07	0.08	0.32	0.05	0.14	0.18	0.64
Vulnerable	0.04	0.02	0.01	0.04	0.10	0.28	0.08	0.14	0.31	0.61
Middle income	0.03	0.01	0.02	0.03	0.06	0.23	0.06	0.22	0.36	0.59
High income	0.01	0.03	0.05	0.01	0.01	0.20	0.05	0.16	0.49	0.50

Source: Olifants environmental SAM, 2012

5.3.1 Aggregate Impacts of the Water Pollution Tax

We provide here a brief discussion of the aggregate impact of the water pollution tax. Table 5.3 shows the impacts of the pollution tax on total nitrogen discharged, aggregate welfare[15], as well as factor returns in the three policy scenarios. The result indicates that the pollution tax achieves its purpose of reducing nitrogen emissions by shifting production away from pollution-intensive sectors. Aggregate welfare which is measured by the Hicksian equivalent variation decreased under the no-revenue recycling scenario compared with the revenue recycling scenarios. Of the two revenue recycling scenarios, uniform government transfers to households outperform production subsidy to pollution abatement sectors. The reason is that the uniform government transfer which is a direct subsidy to households' boosts their income and enhances demand for consumption goods via higher purchasing power. On the other hand, the subsidy to pollution abatement sectors reduces the cost of their production, thus boosting the capacity of the regional economy to clean up. It should, however, be noted that losses in household welfare only reflect changes in consumption because the benefits of water quality improvements are not considered in their utility function. Our welfare results can, therefore, be considered as a lower bound on the welfare gains from improving water quality in the Olifants river basin.

Table 5.3: Aggregate Impact of a 50% Increase in Pollution Tax on Nitrogen Emission under Alternative Revenue Recycling Scenarios (in %age Change)

	No-revenue recycling	Uniform transfers to households	Production subsidy to pollution abatement sectors
Total nitrogen discharged	-0.330	-0.260	-0.210
Welfare	-0.058	0.044	-0.006
Wage rate highly skilled labour	-0.210	-0.153	-0.070
Wage rate skilled labour	-0.263	-0.188	-0.081
Capital rental price	-0.319	-0.216	-0.098

Source: Olifants environmental CGE model

15 Aggregate welfare is a simple sum of the welfare results for all household groups. That is, we did not apply a weighting scheme.

5.3.2 Distributional Impacts and Remedial Policy Options

Without Revenue Recycling

The distributional impact of the water pollution tax on households differs as a result of differences in spending and income patterns. The pollution tax affects the relative prices of pollution-intensive and non-polluting goods (spending side) as well as the returns to capital and labour (income side). The impact on factor returns depends on the factor intensity of each sector thus; the relative contribution of the different factor endowments on household income drives the distributional results from the income side.

The immediate impact of introducing the pollution tax is an increase in the marginal cost of production in polluting sectors, causing them to re-optimize to lower output levels. This negatively impacts demand for primary factors, particularly capital, since polluting firms are relatively capital intensive. Although, prices of both capital and labour fall, capital bears the bigger burden of the tax increases (see Table 5.3), because the reduction in capital by polluting firms outweighs the reduction in labour. All households experience a fall in income due to lower remuneration to factors of production. However, wealthy households experience a larger fall in income because of a bigger fall in the relative return to capital and the fact that they derive a greater share of their income from capital endowments (see Table 5.1). Poor households, on the other hand, receive a greater share of their income from labour, mainly unskilled labour and transfers which is fixed because it is indexed to inflation. Although the impact of the tax on the vulnerable group is also high, the share of total income for both poorest and vulnerable groups increases after the tax (see column 3 of Table 5.4). Therefore, the distributional impact of the pollution tax is progressive from the income side (i.e. poor households are less affected by the changes in factor prices).

Table 5.4: Impact of the Water Pollution Tax on Income, Consumer Spending, and Net Income (without revenue recycling)

Households	Share in total income before tax	Share in total income after tax	%age change in income after tax	Equivalent variation	Net income
Poorest	0.141	0.151	-0.167	-0.021	-2.662
Vulnerable	0.150	0.154	-0.204	-0.015	-1.665
Middle income	0.159	0.158	-0.227	-0.012	-1.662
High income	0.551	0.537	-0.243	-0.010	-1.138

Source: Olifants environmental CGE model

From the consumption point of view, the imposition of the pollution tax raises the price of pollution-intensive goods causing households to reduce the share of pollution-intensive goods in their consumption basket (i.e. demand for pollution-intensive goods fall by a greater percentage compared with non-polluting goods). However, the increase in the prices of pollution-intensive goods adversely affects the poor than the rich, because the former allocate a greater share of their expenditure on pollution-intensive goods (see Table 5.2). We use the Hicksian equivalent variation[16] calculated for each household as a welfare indicator to assess the distributional impact of the pollution tax from the spending side. Column 5 of Table 5.4 shows that poor households would be willing to pay twice as much as rich households to avoid the implementation of the water pollution tax policy. This suggests that from the spending side, the pollution tax is not pro-poor because it hurts the poor more than the rich (i.e. it's regressive).

Given that the pollution tax is poverty reducing from the income side and poverty increasing from the expenditure side, we assess its net impact using net income (i.e. the difference between income and expenditure). The last column of Table 5.4 indicates that the negative impact on the expenditure side more than offset the positive impact on the income side. As a result, the introduction of a water pollution tax in the Olifants river basin without revenue recycling will hurt the poor. Our result is consistent with the literature on the distributional impact of carbon/energy taxes which argue that the distributional impact of these taxes is due to the higher pollution-intensive or energy expenditure share of poor households.

Under Revenue Recycling

In this subsection, we discuss the results of two revenue recycling options employed to offset the negative impact of the tax policy on the poor. Overall, recycling the pollution tax revenue improves the welfare of all households with the poor gaining more from the redistribution. For instance, under the government transfers to households recycling option, poor households would be willing to pay at most R 78,000 to see the implementation of the pollution tax policy whereas rich households would be willing to pay a maximum amount of R 105,000 to avoid the lower utility level due to the pollution tax policy (see column 5 of Table 5.5). In addition, column 2 of Table 5.5 indicates that poor households increase their share in total income after the introduction of the pollution tax relative to rich households. Thus, recycling the tax revenue will have a positive impact on equity by redistributing income from the

[16] The formula used was adapted from Blonigen et al. (1997).

better-off to the poor and vulnerable. The lower impact of the production subsidy recycling option is caused by the restrictive specification of the fixed coefficients Leontief production technology, which does not allow substitution flexibility between inputs. Production activities in our model buy abatement for only a fixed ratio of total pollution generated and the remainder is disposed of in the environment, making the supply of pollutants move in direct proportion to the level of economic activity.

Table 5.5: Impact of the Pollution Tax on Income, Consumer Spending, and Net Income (with revenue recycling)

Households	Share in total income before tax	Share in total income after tax	%age change in income after tax	Equivalent variation	Net income
Uniform Transfers to Households					
Poorest	0.141	0.195	0.423	0.078	0.150
Vulnerable	0.150	0.179	0.219	0.038	0.094
Middle income	0.159	0.184	0.183	0.035	0.094
High income	0.551	0.443	-0.177	-0.105	0.064
Production Subsidy to Pollution Abatement Sectors					
Poorest	0.141	0.146	-0.026	-0.005	-0.357
Vulnerable	0.150	0.153	-0.043	-0.004	-0.521
Middle income	0.159	0.159	-0.063	-0.007	-0.522
High income	0.551	0.542	-0.077	-0.011	-0.835

Source: Olifants environmental CGE model

5.4 **Summary**

This chapter used the Olifants environmental CGE model to assess the distributional impact of a water pollution tax considering both the income and spending-side effects. The analysis contributes to the literature on the distributional impacts of environmental policies which have been dominated by the incidence of carbon and energy taxes. The chapter also analysed the distributional impact of two revenue recycling schemes to mitigate the potential adverse effect of the policy. The simulation results indicate that without revenue recycling, the water pollution tax is progressive from the household income perspective but regressive from the household spending perspective. The net impact is, however, regressive showing that the spending-side effect dominates the income-side effect. This finding stems from the large expenditure share of pollution-intensive goods in the consumption basket of poor households compared to the rich in the basin. Depending on the revenue recycling scheme, the regressive effect of the pollution tax is either weakened or reversed. Recycling the tax revenue through government transfers to households leads to progressive welfare outcomes, whereas returning the tax revenue to pollution abatement sectors in the form of a production subsidy weakens the regressive effect. The weak impact of the production subsidy recycling option is due to rigidities on the production side of the model. Our finding is, however, consistent with the literature and highlights the need to analyse the distributional impacts of environmental policies from both the income and spending-sides.

CHAPTER 6: SUMMARY, CONCLUSIONS, AND IMPLICATIONS FOR RESEARCH AND POLICY

6.1 Introduction

This chapter provides a summary of the main findings of this thesis and the implications for research and policy. Section 6.2 summarises the thesis with conclusions presented in section 6.3. The implications of the results for policy and research are presented in section 6.4 while section 6.5 provides the limitations of the study and areas for further research.

6.2 Summary of Thesis

The research presented in this thesis had two main objectives. The first was to integrate water pollution and abatement measures into a computable general equilibrium (CGE) model to account for both the direct and indirect costs of water quality management policies. The second was to apply the model to investigate the environmental, economic, and social impacts of water quality management policies using the Olifants river basin as a case study.

Chapter one provides a motivation for the research. It began with the water scarcity and quality problems in South Africa in general and the Olifants river basin in particular. To improve the deteriorating water quality situation, the government has intended to enhance enforcement of available water quality management policies to protect the country's water resources for ecological sustainability and socioeconomic growth. However, unnecessarily strict regulation could have adverse economic and social impacts thus, there is a need to analyse the trade-offs (social, economic, and environmental) associated with the implementation of these policies. In this regard, the chapter suggested an integrated modelling approach that can account for both the direct and indirect effects as well as quantify the magnitude of the various effects of the policies.

In chapter two, the relevant literature on analytical approaches and empirical methods employed to study the social and economic implications of environmental policies and related studies were reviewed. First, the chapter reviewed partial equilibrium models which portray behavioural relations that underpin outcomes in a single market by tracking the effects of policy changes such as pollution control on production and consumption decisions. Even though they have the advantage of modelling in detail the supply and demand of the commodity under analysis, they fail to account for spillover effects on other commodities and factor markets which become vital because environmental policy reforms lead to non-marginal changes in the economy. Second, economy-wide modelling approaches that overcome the limitation of partial equilibrium models were reviewed. These approaches which include input-output, social accounting matrix based and computable general equilibrium models account for the interdependencies that exist between different markets and show that changes in government policy have far-reaching impacts on the entire economy and are not confined to a single market. The chapter suggested that the CGE approach with its appeal of endogenous price determination and substitution possibilities in supply and demand systems is capable of more accurately simulating the results of policy changes than the other approaches. Although the approach has been extensively used to analyse the economic costs and effectiveness of environmental management policies in South Africa, the focus has been on managing the quantity dimension of water with no effort so far made to analyse water quality management dimensions using the CGE framework.

To motivate the need for an analytical framework that integrates both economic and environmental activities, chapter three began with a brief overview of the interaction between an economy and its environment. The chapter adapted a standard static neoclassical CGE model to the requirements of a regional model and integrated an environmental component. The CGE model covers the indirect economic costs of the water quality management policy while the environmental component describes the direct environmental costs. The environmental component includes information on water pollution and abatement activities of production sectors, pollution taxes, and subsidies. In an economy with water quality management policies, polluters are liable to pay for their pollution discharges. Thus, two types of pollution-related costs were clearly specified in the model: water pollution taxes and pollution abatement costs. Water pollution taxes are collected by the government and the amount paid by a polluter depends on its output, pollution emission tax rates, pollution intensities, and pollution clean-up rates. Pollution abatement costs are also defined by a polluter’s output, the price of pollution abatement services, pollution intensities, and the proportion of total pollution abated. Pollution abatement is a service provided by incorporated pollution abatement sectors whose sole mandate is to provide the best available cleaning or purification services to help polluters meet prescribed environmental standards. The demand and price of pollution abatement services are endogenously determined in the model based on prevailing market conditions.

To calibrate the Olifants environmental CGE model, chapter three also constructed an environmental SAM database for the basin using the framework of an environmentally extended SAM developed by Xie (2000). The environmental SAM includes three water pollutants (salinity, nitrogen, and phosphorus) and distinguishes production activities from pollution abatement activities whose output are treated as special intermediate inputs in the commodity account. In addition, the environmental SAM has three pollution tax accounts which receive payments from production sectors for their water pollution discharges. The environmental SAM which links economic and water pollution and abatement activities serve as a consistent database for calibrating the Olifants environmental CGE model. The calibration approach was adopted to estimate model parameters due to its simplicity and limited data requirements.

In chapters four and five, the Olifants environmental CGE model was applied to assess the environmental, economic and distributional impacts of taxing water pollution in the Olifants river basin. Chapter four was devoted to analysing the trade-offs between economic growth and the protection of environmental quality whereas chapter five investigated the impacts of the tax policy on the different social groups in the basin. Both chapters four and five also analysed the potential mitigation impact of two revenue recycling schemes. A range of sensitivity analysis was performed to test the robustness of the results to changes in parameter values and assumptions.

6.3 Conclusions from Policy Simulations

The policy scenarios ran in this study assumed arbitrarily that the government raises the pollution tax rate on nutrient load (total nitrogen and total phosphorus) by 50% with reference to the base value with the intention of reducing the total nutrient load in the Olifants River. The policy simulations provide rich information about the environmental, economic, and social impacts of water quality management interventions in the basin.

The results show that internalising the negative externality of water pollution in the Olifants river basin will effectively achieve its environmental objective of protecting water quality through shifting production away from pollution-intensive sectors. Environmental protection is, however, achieved at a cost to economic growth (i.e. loss in real regional GDP) and households' welfare. The economic and social burden of the pollution tax happens to be, however, insignificant due to the small relative share of the water pollution supply and abatement costs in total production costs. Also, the impacts were found to be small regardless of the elasticities of substitution in production and trade used in the analysis. The sectors with the highest impacts on wastewater discharge in the basin were found to be manufacturing (primarily food processing and other manufacturing activities), agriculture (especially horticulture and livestock sectors), and mining. Together, these sectors generate over 90% of the pollution in the basin but they also generate the bulk of the Olifants' export earnings. In terms of the distributional impacts, the results indicate that the water pollution tax is progressive (inequity and poverty-reducing) on the income side as the poorest and vulnerable derive lower shares of their income from capital, which bears the biggest burden of the tax. On the expenditure side, however, the tax is regressive (inequity and poverty increasing), due to the higher share of pollution-intensive goods in poor households' expenditure. The net effect of the tax policy is, however, not pro-poor.

Results from the alternative revenue recycling scenarios suggest a high potential for fiscal policy regimes to mitigate the economic and social burden of the tax. In general, recycling the tax revenue through direct government transfers to households' outperformed a subsidy to pollution abatement sectors mainly due to rigidities on the production side of the model.

From the above findings and within the context of our data and analysis, we conclude that the implementation and the enhance enforcement of the water pollution control policies in the Olifants river basin will minimise pollution discharge and protect the aquatic ecosystems with minimal impacts on socioeconomic welfare.

6.4 **Implications for Research and Policy**

This study has provided quantitative estimates of the trade-offs between protecting water resources and the economic and social burden of water quality management interventions in the Olifants river basin. The results are useful to national and regional water managers and policy makers to understand the quantitative impact of these policies on the environment, economy, and social welfare. The study has also demonstrated the importance of assessing the impact of these policies using an integrated framework that links economic activities and water pollution and abatement measures to capture both the direct and indirect costs.

An important finding of this study is that the economic and social costs of implementing water quality management policies in the Olifants river basin is fairly small even when the tax revenue is used for fiscal adjustment by the government (i.e. the revenue is not disbursed back to the basin to address water quality problems or used to improve the living conditions of the basin's population). This is because of the small relative share of the water pollution supply and abatement costs in total production costs; implying that a large tax rate on water pollution will have little impact on economic and social activities and can obtain larger water quality dividend. This finding is instructive given the restrictive assumptions in modelling the production structure of pollution abatement supply and demand in this study. Furthermore, the finding suggests that allowing flexibility on the production side of the model (i.e. improving the flexibility with which polluters can reduce their tax liability) would yield even much lower economic and social costs.

This research has also highlighted the relevance of assessing the social costs of environmental management policies in a general equilibrium setting that allows non-linear substitution possibilities and endogenous price determination. For the reason that the implementation of these policies not only change prices of pollution-intensive commodities but also factor prices due to their impact on the cost of production. As a result, a proper assessment of the social burdens of these measures should simultaneously account for the impact on households' income and spending.

To the extent that taxing water pollution will raise revenue for the government, this research has also shown that revenue recycling has an important implication on both the efficiency and equity of environmental management policies. Thus, analysis of these policies without considering the use of the revenue can lead to misleading policy prescriptions.

6.5 **Limitations of the Study and Areas for Further Research**

Although this research has provided an empirical understanding of the interactions between economic variables and water pollution and abatement measures in the context of water quality management policies, it has some limitations inherent in its basic assumptions, particularly in modelling the production structure of pollution abatement supply and use activities. Demand for pollution abatement services by production sectors, for instance, is specified in a simple way using exogenously determined clean-up rates and the assumption that unit costs of pollution abatement are fixed. Future research efforts should aim to relax these rigid assumptions as better data become available for more realistic specification of abatement demand functions and endogenously determined clean-up rates.

The analysis also does not account for the economic benefits from water quality improvements as well as from technological advancements that lead to reduced pollution intensities. To this extent, the estimated economic costs required to reach the prescribed water quality standards should be considered as upper bounds. Also, the estimated welfare results should be considered as a lower bound on the welfare gains from improving water quality in the Olifants river basin. Moreover, in a dynamic setting, the policy will influence polluters' investment decisions such as investing in 'in-house' wastewater treatment plants that reduce pollution discharge and improves output. This will lower the economic and social burden of the policy and also provide insights into the path required to meet prescribed policy standards.

Furthermore, as often argued in the distributional impact literature; a comprehensive study of poverty and inequality lies in micro data. In this context, the micro-simulation approach where actual individuals or households from surveys are employed provides rich information (such as intra-group variations) compared to the representative household assumption which only allows inter group comparisons. Also, as mentioned in chapter four, this study did not cover pollution from municipal waste though is a major water pollution source in the study area.

Though the use of a single test pollutant was valid for illustration, it would have been interesting to investigate the combined effects of a simultaneous imposition of the tax on multiple pollutants especially considering that changes in economic activity will impact the emission of several pollutants. Future research efforts should also consider the simultaneous impact of the tax on multiple pollutants to account for the pollutant interaction effect.

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APPENDIX

Table A1 | Micro and Macro Level Impacts of a 50% Increase in Pollution Tax on Phosphorus Emission under Alternative Revenue Recycling Scenarios (%age Change Relative to Base-run)

	Base-run (units)[a]	**Scenario 1: No-revenue recycling (%age change)**	**Scenario 2: Uniform transfers to households (%age change)**	**Scenario 3: Production subsidy to pollution abatement sectors (%age change)**
Total phosphorus discharged	1.515	−1.32	−1.04	−1.09
Changes in sectoral output				
Field crops	3.273	0.53	1.69	0.41
Horticultural crops	6.557	−5.82	−5.51	−4.69
Livestock	4.093	−1.29	−1.23	−1.03
Other agriculture	5.975	−0.66	0.06	−0.59
Mining	78.186	−0.28	−0.17	−0.18
Chemical manufacturing	4.478	−5.54	−4.45	−4.74
Food, beverage and tobacco	35.997	−1.52	−1.10	−1.32
Wood and paper	7.124	−2.78	−2.26	−2.38
Other manufacturing	49.378	−0.90	−0.74	−0.74
Services	157.614	0.75	0.89	0.60
Changes in macro-aggregates				
RRGDP	214.947	−0.11	0.08	−0.09
Private consumption	125.623	−0.19	0.13	−0.16
Exports	102.360	−0.35	−0.17	−0.27
Imports	96.357	−0.37	−0.18	−0.28
Real exchange rate	1.0	−0.011	0.024	−0.023
Unskilled labour employment	18.813	−0.91	−1.34	−0.75
Government revenue	46.876	2.47	2.98	2.02
Aggregate government transfer	1.147	0	149.92	0
Government savings (surplus)	−1.354	−108.59	0	−89.27
Aggregate household savings[b]	25.689	−5.05	0.31	−4.16
Total absorption	208.944	−0.12	0.08	−0.09

See notes to Tables 4.2 and 4.3

Table A2 | Micro and Macro Level Impacts of a 50% Increase in Pollution Tax on Salinity under Alternative Revenue Recycling Scenarios (%age Change Relative to Base-run)

	Base-run (units)[a]	**Scenario 1: No-revenue recycling (%age change)**	**Scenario 2: Uniform transfers to households (%age change)**	**Scenario 3: Production subsidy to pollution abatement sectors (%age change)**
Total dissolved salts discharged	2.153	−0.24	−0.19	−0.15
Changes in sectoral output				
Field crops	3.273	0.15	0.34	0.10
Horticultural crops	6.557	−1.45	−1.39	−0.99
Livestock	4.093	−0.28	−0.26	−0.18
Other agriculture	5.975	0.004	0.13	0.03
Mining	78.186	−0.21	−0.19	−0.17
Chemical manufacturing	4.478	−0.64	−0.45	−0.32
Food, beverage and tobacco	35.997	−0.06	0.02	0.02
Wood and paper	7.124	−0.34	−0.23	−0.18
Other manufacturing	49.378	0.07	0.10	0.12
Services	157.614	0.14	0.18	0.09
Changes in macro-aggregates				
RRGDP	214.947	−0.02	0.01	−0.01
Private consumption	125.623	−0.03	0.04	−0.02
Exports	102.360	−0.12	−0.04	−0.08
Imports	96.357	−0.12	−0.04	−0.09
Real exchange rate	1.0	0.037	0.044	0.033
Unskilled labour employment	18.813	−0.12	−0.24	−0.06
Government revenue	46.876	0.39	0.48	0.22
Aggregate government transfer	1.147	0	23.12	0
Government savings (surplus)	−1.354	−16.37	0	−8.91
Aggregate household savings[b]	25.689	−0.71	0.08	−0.37
Total absorption	208.944	−0.02	0.01	−0.01

See notes to Tables 4.2 and 4.3.