A COMPARISON OF THE COST ASSOCIATED WITH POLLUTION PREVENTION MEASURES TO THAT REQUIRED TO TREAT POLLUTED WATER RESOURCES

Report to the
Water Research Commission

by

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EXECUTIVE SUMMARY

Background and motivation

Pollution of water resources is largely as a result of human activity and therefore could be prevented to a large degree. The quality of our water resources is deteriorating and downstream water users have to deal with the pollution impacts caused by upstream uses. This situation have resulted in a debate as to whether it will not make more economic sense for water users to treat water for use rather than to meet discharge standards. There are instances where discharge standards require water of a better quality to be released than what was abstracted for use in the first place. However, the National Water Act, 1998 places the responsibility for pollution prevention and remediation on the polluter, not the user. The development of the Waste Discharge Charge System (WDCS) is aimed at providing economic incentives and penalties for polluters to pay in accordance with the "polluter-pays-principle".

The hypothesis that pollution prevention is preferable as compared to a pollution clean-up regime, but also that unnecessarily strict regulation will have negative consequences for the economy, was tested. It was therefore necessary to determine the cost of pollution prevention as well as the cost of pollution to downstream users. Four prominent water pollution issues, namely salinisation, nutrient enrichment, microbial degradation and sediment migration, have been identified as the focus of this research which was undertaken as a solicited project funded by the Water Research Commission. This study therefore set out to identify the sources of the pollution in the identified study areas. The cost impacts of each of the pollution issues to downstream water users were determined where possible. This cost was then compared to the cost of pollution prevention measures.

Objectives of the research

The general objective of the study was to compare the costs associated with measures to control pollution at source with those required to treat the consequences associated with polluted water for salinity, eutrophication, microbial pollution and sediments.

This general objective was divided into five sub-aims namely:

- To identify the major sectors contributing to salinisation, eutrophication, microbial pollution and sedimentation and their position in the South Africa economy;
- To do a cost benefit comparison of water treatment and pollution abatement in an urban/industrial context;
- To do a cost benefit comparison of water treatment and pollution abatement in an agricultural context;
- To determine the current distribution of costs associated with pollution between the different water users; and
- To analyse the transfer of benefits between polluters and users resulting from pollution control at source and the associated economic consequences.

Results and discussion

The results of this study must be read with caution as site-specific information based on a variety of critical assumptions were used to get to answers. Therefore, figures provided should be interpreted as an indication of the order of magnitude of the costs incurred. Although the results are site-specific and based on assumptions, it is good enough to inform policy.

Waste water treatment plants (WWTW) are the main line of defense to prevent pollution from urban and industrial areas. The success of this prevention measure is therefore highly dependent on adequate capacity to deal with the volumes and quality of effluent as well as proper operations at these facilities. The general poor state of affairs at the majority of these facilities as indicated by their Green Drop Status is therefore a major cause for concern. The calculations in this report did not consider compliance to discharge standards. It should also be noted that economies of scale play a huge role in the cost of water purification and waste water treatment, therefore average values were used in the cost calculations.

The cost of nutrient removal technologies at WWTW were used as a proxy for eutrophication prevention while the infrastructure and technology upgrades at the Rietvlei water purification works combined with the rehabilitation cost for the Hartbeespoort Dam provided insights into the cost impacts of eutrophication.

Pollution from urban and industrial areas that is not intercepted by the sewerage network enters the water resource as diffuse sources of pollution. These sources are by nature difficult to trace. It is also largely impossible to allocate liability for diffuse pollution in terms of the 'polluter pays principle'. The cost of implementing pollution prevention measures to reduce diffuse sources of pollution from urban areas was not considered.

Water used for irrigated agriculture is generally not treated before use. This means that crop yields are increasingly at risk as a result of increased pollution. In this regard, the cost impacts of increased salinity levels on the total gross margin above specific cost for commercial irrigation in the Loskop WUA were determined. Mines were identified as a major source of salinity in the study area. The cost of collection and treatment of mine effluent in the Olifants WMA to a quality suitable for irrigation purposes was therefore used for the cost comparison.

The impacts of microbial pollution are difficult to determine, other than to look at burden of disease as a result of microbial pollution and the associated opportunity costs. In South Africa 84% of all diarrhoea incidences are attributable to water and sanitation. Since sanitation services are the prevention measure applicable to microbial pollution, the costs for providing improved sanitation services in areas with below RDP service levels in the Olifants WMA was used in the calculations.

Sedimentation is especially problematic in dams where it impacts on storage capacity and potentially long term water security. The cost of sedimentation was calculated as a loss in storage capacity. However, it was not possible to put a monetary value to sediment control measures in the study area.
Numerous challenges hinder a fully inclusive and accurate valuation of the costs associated with water pollution. However, the absolute quantification of the burden is not necessary as the central point is clear: water pollution poses a serious threat to human health in the study area. What is the price of a human life?

Pollution is by nature a social burden and the cost associated with this burden is clearly too high to be sustainable into the future. In addition, the pollution loads in our rivers is increasingly putting pressure on water purification systems and the associated costs are borne by the water users. This situation is unacceptable as it is in conflict with the 'polluter pays principle'.

Where a direct cost comparison was possible the answer is clear, it is indeed cheaper and better to prevent pollution than to treat the consequences of pollution.

Recommendations

This study clearly indicated the benefits of access to safe water supply and improved sanitation services. It is therefore recommended that improved access to these services and hygiene must be prioritised. Diarrhoea morbidity and resultant direct treatment and transport costs in the Olifants WMA could be drastically decreased if more people gained access to these services.

Although it is clear that pollution prevention is better than cure and cheaper, it is clear that water quality is deteriorating over time and distance in the study areas. To ensure continued fitness for use of these resources, the pollution levels must at least be stabilised at current levels before reaching a point of no return for treatment to ensure continued fitness for use. Introduction and enforcement of stricter pollution prevention measures combined with pollution abatement strategies are required.

It was also very clear that the 'polluter-pays-principle' is not applied and enforced in the study areas. The external costs associated with water pollution are costing the country thousands of millions while water security is also at stake. Implementation of the 'polluter-pays-principle' also requires better policing.

Finally, a water pollution audit on WMA level is proposed to form the baseline for the enforcement of the polluter-pays-principle' as a partial solution. Implementation of the Waste Discharge Charge System should be speeded up to ensure that cost of pollution is internalized.

Knowledge dissemination

Some of the findings of the research on which this report is based were presented at WISA 2010 and the CSIR's Science Real and Relevant Conference 2010. It is envisaged that at least two journal papers (one in preparation) will be published on this work.

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ABBREVIATIONS

AIDS	Acquired Immune Deficiency Syndrome
AMD	Acid Mine Drainage
BoD	Burden of Disease
COD	Chemical Oxygen Demand
COI	Cost of Illness
CPI	Consumer Price Index
CSIR	Council for Scientific and Industrial Research
CVM	Contingent Valuation Method
DAF	Dissolved Air Flotation
DALY	Disability Adjusted Life Year
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
EC	External Cost (not to be confused with electrical conductivity)
EMF	Environmental Management Framework
ERWAT	East Rand Water
GBD	Global Burden of Disease
GDP	Gross Domestic Product
GIS	Geographical Information System
HIV	Human Immunodeficiency Virus
IWRM	Integrated water Resource Management
KNP	Kruger National Park
LP	Linear Programming
MVP	Marginal Value Product
ORT	Oral rehydration Therapy
PC	Private Cost
RDP	Reconstruction and Development Programme
RSA	Republic of South Africa
StatsSA	Statistics South Africa
TDS	Total Dissolved Salts
TGMASC	Total Gross Margin Above Specific Cost
TOR	Terms of Reference
TSS	Total Suspended Solids
VIP	Ventilated Improved Pit Latrine
WDCS	Waste Discharge Charge System
WMA	Water Management Area
WPW	Water Purification Works
WRC	Water Research Commission
WTP	Willingness to Pay
WWTW	Waste Water Treatment Works
WUA	Water User Association
YLD	Years Lived with Disability
YLL	Years of Life Lost
ZAR	South African Rand

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1 Background

This project was initiated as a solicited call for proposals by the Water Research Commission (WRC) and the CSIR was appointed to undertake the research over a period of three years.

The terms of reference (ToR) for this project hypothesized that pollution prevention is preferable as compared to a pollution clean-up regime, but also that unnecessarily strict regulation will have negative consequences for the economy. The White Paper on Integrated Pollution and Waste Management for South Africa (DEAT, 2000) identifies nutrient enrichment, salinisation, microbial degradation and sediment migration as prominent water pollution issues that need to be addressed. Surface water resources provide most of the country's water requirements for urban, industrial and irrigation purposes (DWAF, 2004a) while in terms of the law, seepage, runoff or water containing waste, which emanates from the use of water, must be returned to the water resource from which it was taken, unless the responsible authority directs otherwise or the relevant authorisation provides otherwise (RSA, 1998: Clause 22(2) (e)). Pollution of water resources could therefore largely be ascribed to human activity and to a lesser degree to natural causes such as the underlying geology and through natural erosion caused by wind and water run-off. In order to manage pollution from anthropogenic sources, the national water policy for South Africa (DWAF, 1997) set as a principle that water quality management options shall include the use of economic incentives. The Department of Water Affairs has already started the development of the Waste Discharge Charge System (WDCS) aimed at providing economic incentives and penalties to operationalise the "polluter pays principle" (Goodstein, 2008, Glazyrina *et al*., 2006, Taviv *et al*., 1999), as adopted by both of the mentioned policies. Therefore, the level at which the charges will be set, will have to be a reflection of the actual cost resulting from the polluted water resources.

This study was not intended to set the level of these charges, but aimed to put a rand value to direct and indirect costs associated with pollution of water resources in the indentified study areas. It was decided to investigate the four most problematic water quality problems experienced in South Africa.

1.1 Aims

The general aim of the study was to compare the costs associated with measures to control pollution at source with those required to treat the consequences associated with polluted water for salinity, eutrophication, microbial pollution and sediments.

This general aim was divided into five sub-aims namely:

- To identify the major sectors contributing to salinisation, eutrophication, microbial pollution and sedimentation and their position in the South Africa economy;
- To do a cost benefit comparison of water treatment and pollution abatement in an urban/industrial context;
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- To analyse the transfer of benefits between polluters and users resulting from pollution control at source and the associated economic consequences.

1.2 Approach

A differential approach was followed in conducting this project to provide for the differences in the sources of the contaminants investigated and the levels of information available for each. Custom-made approaches tailored to each of the four contaminants were adopted. While the cost associated with salinisation of water resources has (for example) been determined in an earlier WRC project entitled "Negative economic effects of salinity" (report TT123/00) and eutrophication costs was being determined by Umgeni Water in a WRC project entitled "Development of a model to assess the costs associated with eutrophication", the first-order costs associated with microbial pollution and sedimentation in the identified study areas was determined as part of this project. The approach adopted for the assessment of the four contaminants varied, but to ensure that the level of assessment and the reported results are comparable, the same tools e.g. cost benefit analysis, was used throughout.

A desktop study was undertaken to identify catchment areas where nutrient enrichment, salinity, microbial pollution and sedimentation are known to be problematic and from which data could be obtained with relative ease. The next step was to identify the potential sources and impacts of pollution on downstream users. The majority of urban water users including industry obtain water from municipal distribution systems at potable quality. It was therefore decided not to do a user-specific study, but to focus on:

- Potable water production;
- Wastewater treatment;
- Commercial agriculture;
- Human health; and
- Loss of storage capacity, within a water management area.

It was also necessary to understand the economics of water pollution and the value of water as an economic good.

2 Overview of each pollutant

Before discussing the pollution sources and impacts in the identified case study areas, a brief overview of each pollutant is given. The background levels that can be expected in un-impacted resources are provided where available. It is however important to note that – for aquatic ecosystems – the guideline values quoted in tables 2 and 5 refers to any change that might take place due to the discharge of an effluent containing pollutants. It is therefore necessary to read the 'guideline' in conjunction with monitoring data. The quality guidelines for use (especially drinking water) refer to the quality of water after treatment.

2.1 Eutrophication and its impacts

Eutrophication is defined as enrichment of aquatic ecosystems by nitrogen and particularly phosphorous (Hart, 2006). The main sources of nutrient pollution are sewage effluent and agricultural run-off. The discharge of nitrogen (as nitrates) and phosphorus (as phosphates) to inland rivers, lakes and dams causes massive growth of algae and plants due to the "fertilizer type" effect of the phosphate and nitrate (Steyn *et al.*, 1975). Phosphate is a more limiting factor than nitrate in eutrophication, because some bacteria and algae are able to fix atmospheric nitrogen (Correll, 1998).

Eutrophication is therefore a process of nutrient enrichment of a system that can be, and is, used to classify the stage at which this process is at any given time in a particular water body (DWAF, 2003b). The 'trophic status' of a water body is therefore a description of the water quality status of that specific water body in terms of nutrient enrichment. A water resource can be classified into one of four classes based on their trophic status (DWAF, 2003b) namely:

1. Oligotrophic – low in nutrients and not productive in terms of aquatic and animal plant life;
2. Mesotrophic – intermediate levels of nutrients, fairly productive in terms of aquatic animal and plant life and showing emerging signs of water quality problems;
3. Eutrophic – rich in nutrients, very productive in terms of aquatic animal and plant life and showing increasing signs of water quality problems; and
4. Hypertrophic – very high nutrient concentrations where plant growth is determined by physical factors. Water quality problems are serious and can be continuous.

In 2003, 76 impoundments were monitored for eutrophication purposes. At that time, eight dams were classified as hypertrophic, six as eutrophic and twelve as mesotrophic (DWAF, 2003b). The distribution of recorded incidences of cyanobacterial toxicity in South African water resources which caused animal death is indicated in Figure 1.

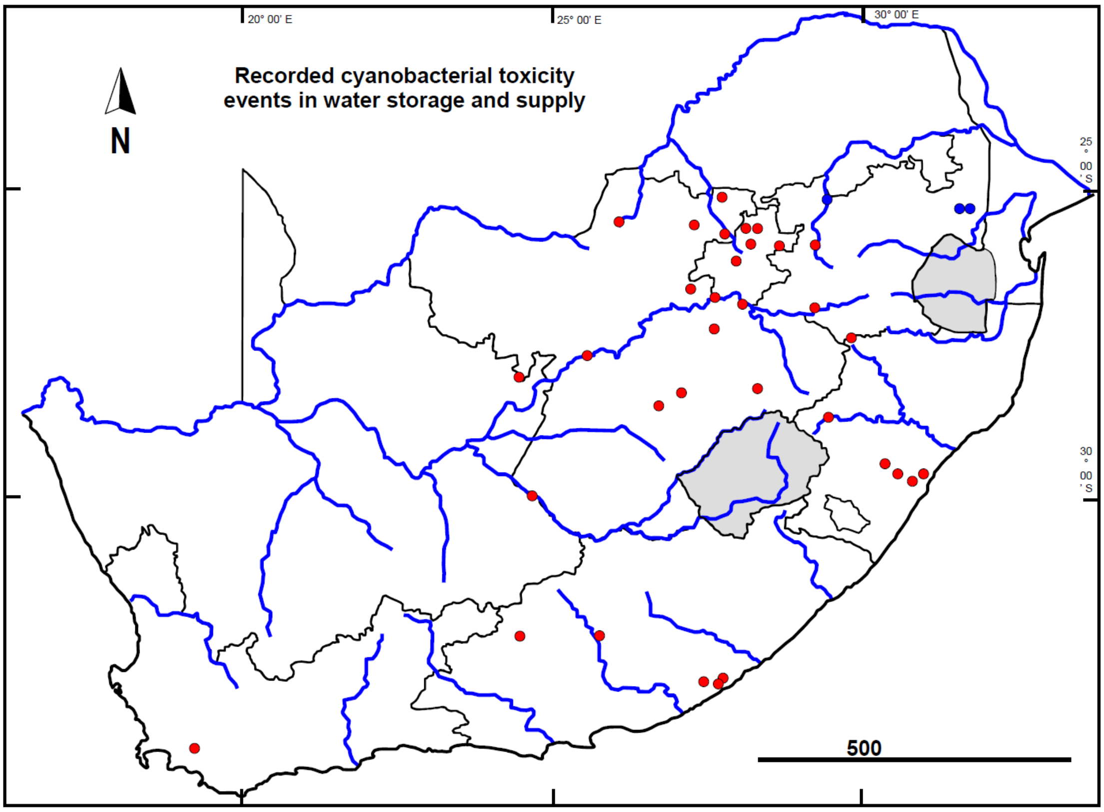

Figure 1: The distribution of recorded cyanobacterial toxicity events in South African water supply reservoirs, which caused the death of livestock, wildlife or domestic animals. (Map drawn from data in Van Ginkel, 2004 [red circles] and recent toxicity events in the Kruger National Park and at Loskop Dam [blue circles]) (Adapted from Ashton, 2010).

The frequent toxic algal blooms associated with eutrophication, can result in fish or animal deaths and the induction of gastro-enteritis in humans (Hohls *et al.*, 2002). A survey of the South African situation revealed that "*eutrophication can be regarded as a national water quality problem in that it has severely degraded certain aquatic systems, and also affects the fitness for use of many water resources throughout the country*" (DWAF, 2003b:18). Eutrophication research traditionally focused on nutrient enrichment of impoundments and lakes because these were used as water supplies. However, evidence from studies performed in a wide variety of geographical locations suggests that flowing waters are also sensitive to anthropogenic inputs of N and P (Smith *et al.*, 1999).

Non-impacted streams typically have an inorganic N:P ratio greater than 25-40:1, whilst most impacted (eutrophic or hypertrophic) systems have an N:P ratio of less than 10:1 (DWAF, 1996g). The target water quality guidelines for N and P are as outlined in Table 1. The values for aquatic ecosystem refer to change that might take place due to discharge of effluent containing nutrients**.**

Table 1: Target Water Quality Guidelines relating to N and P (DWAF, 1996a-g)

Constituent	Aquatic ecosystems	Domestic Drinking	Agriculture		
			Livestock	Irrigation	Aquaculture
Nitrate/Nitrite (mg/ℓ)	See Nitrogen (inorganic)	$0\text{-}6^{*\,\&\,**}$	NA	NA	$0\text{-}100^{*}$ $0\text{-}10^{**}$
Nitrogen (inorganic) (mg/ℓ)	≤ 15% Variation from background	NA	NA	NA	NA
Phosphorous (inorganic) (mg/ℓ)	≤ 15% variation from background	NA	NA	NA	NA

* NO_3^- ** NO_2^-

Excessive algal blooms and nuisance growths of invasive aquatic weeds are the most obvious impacts of eutrophication. However, the environmental consequences of excessive nutrient enrichment are far more serious and far reaching than nuisance increases in plant and algal growth alone. The degradation of water resources by eutrophication can result in losses of their component species, as well as losses of the amenities or services that these systems provide. Typical effects of eutrophication on reservoirs include (Smith *et al.*, 1999):

- Increased biomass of freshwater phytoplankton and periphyton;
- Shifts in phytoplankton species composition to taxa that may be toxic (to humans and aquatic wildlife) or inedible (i.e. bloom-forming cyanobacteria) and, in the worst cases, to microbial populations dominated by bacteria;
- Changes in vascular plant production, biomass and species composition;
- Reduced water clarity;
- Decreases in the perceived aesthetic value of the water body;
- Taste, odour, colour and water supply filtration problems;
- Possible health risks in water supplies;
- Elevated pH and dissolved oxygen depletion in the water column, with attendant changes in the solubility of metal ions and changes in the toxicity of ammonia;
- Increased fish production and harvest;
- Shifts in fish species composition towards less desirable species; and
- Increased possibility of fish kills.

Typical effects of eutrophication on river ecosystems include (Smith *et al.*, 1999):

- Increased biomass and changes in species composition of suspended algae and periphyton;
- Reduced water clarity;
- Taste, colour and odour problems;
- Blockages of intake screens and filters at water purification works;
- Fouling of submerged lines and nets;
- Disruption of flocculation and chlorination processes at water purification plants;
- Restriction of swimming and other water based recreation;
- Harmful fluctuations in pH and in dissolved oxygen concentrations;
- Dense algal mats reduce habitat quality for macro-invertebrates and fish spawning; and
- Increased probability of fish kills.

2.2 Salinity and its impacts

According to the Department of Water Affairs (DWA), the quality of South African water resources is deteriorating mainly due to salinisation, and to a lesser extent eutrophication, trace metals and micro-pollutants (DWAF, 2004a). Salinity of water resources is measured as total dissolved solids (TDS) indicating the total quantity of the various inorganic salts dissolved in the water. High levels of TDS are commonly associated with mining and certain industrial discharges. However, domestic wastewater and run-off from cultivated land and urban areas also contribute to salinity. High sulphate concentrations in the surface water bodies are commonly related to mining effluent discharges (DWAF, 2002 and 2003a).

Natural or unpolluted water also contain varying concentrations of TDS as a consequence of the dissolution of minerals in rocks, soils and decomposing material. The TDS of natural waters is therefore largely dependent on the geological formations in the catchment area with which the water was in contact. The typical TDS concentration in natural waters is provided in Table 2, while the water quality guideline values for the different uses are provided in Table 3. Rainwater typically has a TDS concentration of less than 1 mg/ℓ (DWAF, 1996a).

Table 2: TDS of natural water (DWAF, 1996a)

Geological formations in contact with water	TDS (mg/ℓ)
Granite, siliceous sand and well-leached soils	< 30
Precambium shield areas	< 65
Palaeozoic and Mesozoic sedimentary rock formations	195-1100

Table 3: Water Quality guideline values for TDS (DWAF, 1996a-g)

Constituent	Aquatic ecosystems	Domestic Drinking	Industry Category				Agriculture		
			1	2	3	4	Livestock	Irrigation	Aquaculture
Total Dissolved Solids (mg/ℓ)	< 15% variation from background	0-450	0-100	0-200	0-450	0-1600	0-1000* 0-2000** 0-3000***	≤260	< 15% variation from background

* Dairy, pigs and poultry; ** Cattle and horses; *** Sheep

The TDS concentration in river water increases as it moves downstream, due to the continuous addition of salts from natural and anthropogenic sources. The persistence of salts in the environment is a major problem as increased salinity can result in salinisation of irrigated soils impacting on sensitive crops and resulting in diminished crop yields. It also results in increased scale formation and corrosion of water pipes and changes in biotic communities (DEAT, 2006).

Excessively high levels of TDS in water for domestic use may adversely affect plumbing and appliances resulting in increased maintenance and replacement requirements. Bathing and washing in water with high concentrations of TDS may give rise to excessive skin dryness and soap may lather poorly (DWAF, 1996). Water for human consumption containing high levels of TDS can lead to serious human health effects (DWAF, 1996a) including:

- Laxative effects, mainly from sodium sulphate and magnesium sulphate;
- Adverse effects of sodium on certain cardiac patients and hypertension sufferers;
- Effects of sodium on women with toxaemia associated with pregnancy; and
- Some effects on kidney function.

Salinisation of water remains a major contributing factor to soil degradation in commercial irrigated agriculture in South Africa (Aihoon *et al*., 1997, Armour, 2007). Plants have a natural ability to tolerate water of variable quality (Armour and Viljoen, 2000). However, a salinity problem in irrigated agriculture occurs when salts from the applied irrigation water start to accumulate in the crop root zone, which then affects yield in a negative way once a critical threshold is crossed. This situation has a cumulative effect which, if the accumulating salts are not leached, quickly becomes a serious threat in terms of the profitability of the enterprise. To avoid the accumulation of salts to a harmful level, the salts need to be dissolved and removed by applying enough water to allow percolation through the crop root zone (leaching). The specific volume of water needed for effective leaching is called the leaching fraction, which is defined as the fraction of water entering the soil that passes beyond the root zone (Addiscot, 1977). The leaching factor is a function of the soil type and tolerance of the crop. Structural changes to more salt tolerant crops is often not a viable option, which makes the proper management (by means of suitable drainage) of salinity a critically important determinant of profitability in the irrigation sector (Armour, 2007).

The industrial sector also suffers consequences as a result of increased salinity including damage to equipment and structures as a result of corrosion and scaling. Certain ions in solution may interfere with industrial processes or product quality e.g. patchiness in dyeing or deposition of insoluble salts in medical products (DWAF, 1996c).

2.3 Microbial pollution and its impacts

Microbial pollution relates to the presence of bacteriological, viral, protozoan or other biological contamination of water resources. It can result in serious health concerns for people who are dependent on raw river water for domestic use and contact recreation. Microbial pollution of water also poses health concerns for consumers of fresh food produced by irrigation farming. High concentrations of faecal bacteria are associated with untreated or poorly treated sewage effluents (point sources), and urban run-off (non-point sources). In addition, uncontrolled effluent discharges from the dairy, fish processing, poultry and the red meat industries can contribute to the deterioration of the microbial quality of river water.

Large numbers of waste water treatment works (sewage plants) are not operated optimally when compared to national standards and international best practice (DWA, 2009). It is estimated that less than 15% of all generated waste water receives primary or secondary treatment, while less than 2% receives tertiary treatment (Jos *et al*., 2004). In a 2006 survey, covering 51 small and medium sized waste water treatment works country wide, only 4 % were adequately operated (Snyman *et al*., 2006). The downstream users of the poorly treated sewage effluents are at risk. Being exposed to unsafe water ultimately results in ill health, reduced productivity and increased absenteeism from work. While the acceptability of agricultural crops for human consumption may also be compromised.

Due to the high possibility of pathogens being associated with microbial pollution, indicator organisms are used as management tool to indicate faecal pollution. Traditionally the microbial quality of water is determined by measuring the level of indicator organisms present in the water to reflect the degree of faecal pollution, because of the associated health risks. Indicator organisms that are commonly analysed for are: coliforms, faecal coliforms, *Escherichia coli,* Enterococci and bacteriophages. It should be noted that faecal pollution can be of human or animal origin and does not necessarily indicate the presence of pathogens, but rather the potential for pathogens to be present. In short, there is no direct correlation between numbers of any indicator and enteric pathogens (Grabow, 1996).

The intended end use of any water resource determines the maximum number of indicator organisms that could be allowed to be present in the water. The Microbial Target Water Quality Guidelines specified for specific uses (after treatment) is presented in Table 4 (DWAF, 1996a and DWAF, 1996b).

Table 4: Target Microbial Water Quality Guidelines (DWAF, 1996a; DWAF 1996b)

Constituent	**Domestic** Human use	**Recreation**	
		Full contact	Intermediate contact
Coliforms/100 ml	0	0-150	0-1000
Coliphages/100 ml	0-1	0-20	NA
Enteric viruses ($TCID_{50}$/10 ℓ)	<1	0	NA

The occurrence of diarrhoea is one of the most important impacts of microbial pollution. Lewin *et al.* (2007) found that 84% of all diarrhoeal disease in South Africa is attributable to water pollution and poor sanitation services. It is considered to be a common childhood illness, in both developing and developed countries. Although diarrhoea is not a lifethreatening disease, it is estimated that approximately 1.6 million children die each year from diarrhoea in the developing world (WHO, 2004a).

The World Health Organisation (2002) defines diarrhoea as “the passing of three or more liquid stools in a 24-hour period”. Diarrhoea could further be characterised as “acute watery”, “persistent” or “dysentery”. Acute watery diarrhoea has an abrupt onset and lasts less than 14 days and it usually results in varying degrees of dehydration. Persistent diarrhoea (sometimes also referred to as chronic diarrhoea) lasts more than 14 days, which generally results in significant weight loss due to mal-absorption, nutrient losses and other nutritional problems. Persistent diarrhoea, although only accounting for a small percentage of the total number of diarrhoea episodes, is associated with a significant increase in risk of death (WHO, 2004b; Keusch *et al*., 2006). Dysentery is any stool with visible blood, also referred to as “bloody diarrhoea” where the blood is a sign of intestinal damage caused by inflammation in the intestines (WHO, 2004b; WHO, 2005).

Oral rehydration therapy (ORT) is most often applied to treat the loss of fluids and dehydration due to diarrhoea in an effort to maintain or replenish proper levels of hydration in the body (WHO, 2000). Treatment of diarrhoea in children includes correct fluid therapy, correct feeding therapy, appropriate use of antibiotics, no use of anti-diarrhoeals and effective education of the mother or caretaker.

The aetiological agents mostly responsible for the diarrhoeal burden include viral, bacterial and protozoan organisms. Viral agents responsible for diarrhoea are rotavirus, enteric adenoviruses, astroviruses and caliciviruses. The bacterial agents include the pathogenic and enterotoxigenic *Escherichia coli* strains, shigellae, salmonellae, *Vibrio cholerae* and *Campylobacter jejuni*. The most common protozoan parasites are *Giardia lamblia* and cryptosporidium (Martines *et al*., 1993).

Improved sanitation and clean water plays an important role in prevention of diarrhoeal disease (WHO, 2005). Major improvements have been made in the water and sanitation sector in South Africa since 1994. Almost 88% of people in South Africa (both rural and urban areas) have access to an improved source of drinking water (DWAF, 2009; StatsSA, 2007). However, 14 million people all over the country still lack access to an improved sanitation facility.

Since 1997, diarrhoea related deaths in South Africa have increased from 6 536 to almost 40 000 in 2006 (Steyn and Genthe, 2010). It is acknowledged that HIV plays a major role in diarrhoea incidence and deaths but the main cause of the diseases seems to be water and sanitation related. In addition, households affected by HIV and AIDS require greater quantities of water and excellent hygiene to meet the requirements of the chronically ill and to prevent opportunistic infections (Lule *et al*., 2005). Clearly, incomplete water and sanitation coverage and the associated levels of morbidity and mortality are still not widely appreciated or acknowledged.

2.4 Suspended solids and its impacts

Sediment loads in rivers are measured as total suspended solids (TSS). In South Africa, all rivers, except some in the Natal foothills of the Drakensberg and in the south-western Cape, become highly turbid and laden with suspended solids during the rainy season. The major part of suspended materials found in most natural waters is made up of soil particles derived from land surfaces. Erosion of land surfaces by wind and rain is a continuous and natural process. However, changes in land use such as deforestation and conversion to intensive agriculture or soil loss from construction sites also contribute to the suspended solids entering our water resources. In addition, land-use practices such as mining, overgrazing, non-contour ploughing, removal of riparian vegetation and forestry operations accelerate erosion and result in increased loads of suspended solids in rivers (DWAF, 1996g). Suspended solids includes objects of various sizes ranging from suspensions of dust and pollen particles to cellular suspensions as small as 0.1µm through to large organic and inorganic particles (DWAF, 1996g).

The TSS concentration in surface water increases due to rainfall, discharges containing sediment entering the river and re-suspension of previously deposited sediments. As flow rates decrease, the suspended solids settle out of suspension, the rate of which is dependent on particle size and the hydrodynamics of the water body. A natural variation in TSS concentrations of rivers is governed by the hydrology and geomorphology of a particular region. The target water quality guidelines for TSS are outlined in Table 5Table 5.

Table 5: Target Water Quality Guidelines for TSS (DWAF, 1996c, d, e, f)

Constituent	Industry Category				Agriculture		
	1	2	3	4	Livestock	Irrigation	Aquaculture
Suspended Solids (mg/ℓ)	0-3	0-5	0-5	0-25	NA	0-50	<50* < 20000**

*Clear water species
** Turbid water species

Increased sedimentation loads are considered to be the most important impacts of suspended solids. Specific impacts caused by TSS pollution includes aesthetic issues, increasing water treatment costs, declining fisheries resources and serious ecological degradation of the aquatic environment (Bilotta and Brazier, 2008). The main physical impacts of sediment entering surface water resources are a decline in the storage capacity of dams and blocking of irrigation systems. Sedimentation damage to agricultural land resources include the overwash of infertile material, impairment of natural drainage and swamping due to channel aggregation, associated floodplain scour and bank erosion (Braune and Looser, 1989). In addition, suspended inorganic material carries an electrical charge that could result in a number of dissolved substances, including nutrients, trace metal ions and organic biocides to become adsorbed onto the surfaces of these particles. Substances adsorbed to particles are not biologically available, which may be advantageous in the case of toxic trace metal ions and biocides (DWAF, 1996a). Suspended organic solids on the other hand, may decrease the concentration of dissolved oxygen in the water body due to the oxidation of the solids by micro-organisms (DWAF, 1996b).

An increase in the TSS concentration may also lead to a decrease in the water temperature as more heat is reflected from the surface and less absorbed by the water and may impact the aquatic ecosystem (DWAF, 1996g). Sedimentation can also smother habitats for bottom dwelling organisms.

3 Economics of water pollution

3.1 Water as an economic good

Water is not only a scarce resource in South Africa, it is also subjected to high pollution loads. In South Africa, the White Paper on Integrated Pollution and Waste Management (DEAT, 2000), along with the Water Services Act (RSA, 1997) and the National Water Act (RSA, 1998) create a regulatory framework for water pollution. In addition, the philosophy and principles of integrated water resource management (IWRM) have been widely accepted as a broad strategy for sustainable water management (Funke *et al*., 2007). IWRM prioritises economic values, equity considerations, and environmental sustainability as key objectives; and the challenge is therefore to balance conflicting interests among these objectives. Measures are therefore introduced to ensure the most beneficial utilisation of water in the country both from a social and economic perspective. The measures include the "re-allocation of some water from low benefit uses to higher benefit uses over time" (DWAF, 2004:20). Furthermore, the 1992 Dublin Water Principles and the 2002 World Summit on Sustainable Development support the notion of water as an economic good, which should be managed accordingly to promote equity, efficiency and sustainability (United Nations Department of Economic and Social Affairs: Division of Sustainable Development, 2004). It is therefore clear that economic considerations will increasingly inform water management decision-making and allocation.

Economic valuation of water resources allows for trade-offs to be made among competing resource allocation options, and consequently for decisions to be made regarding resource allocation. The value of water derives largely from its utility (fitness for use), or, put more generally, from its importance and contributions to social welfare (in the broader sense, which includes environmental quality). Such value derives from the fact that water often provides predominantly indirect benefits (i.e., plays a supportive role) to economic or ecological processes; and its value is therefore often embodied in the value of economic or ecosystem goods and services. This is where a strong link to pollution is to be found: pollution distorts trade-offs because it directly affects the "fitness for use" of water. Also, given that pollutants decrease the range of potential uses of water, the opportunity cost of such water is lowered, along with its relative value. Accurate and objective valuation therefore directly informs sustainable water resource management (Birol *et al*., 2006). However, a great deal of uncertainty is associated with the value of a basic resource such as water and pollution adds to this uncertainty.

3.2 Water pollution as a negative externality

The natural environment provides two important functions which support economic development, namely:

1. Natural capital, which is combined with human and financial capital to support economic development; and

2. Assimilative capacity; which allows for absorbing waste created by economic development.

'Pollution' is the result of discharges of by-products into water resources that exceed the natural assimilative capacity of the system and render the water unfit for subsequent uses (DWAF, 2004). Pollution often occurs because natural resources and the services derived from them (such as waste disposal services) are typically under-priced and therefore not taken into account in private decision making.

In a free market economy, private (firm and individual) production and consumption decisions are based on private costs and benefits, which are reflected in market prices. According to neoclassical economics, the 'invisible hand' of the market will ensure that these private decisions will lead to socially optimal outcomes, such as optimal levels of production and pollution (Goodstein, 2008). However, markets are subject to market failures, such as externalities, i.e. external (unaccounted for) costs (such as pollution) and benefits (such as education). Thus, market prices often fail to adequately reflect the full social costs and benefits associated with a particular activity, owing to the existence of externalities, implying that individual decisions based on these prices will not be good for society as a whole. Externalities refer to the side effects (positive or negative) of economic activity (production or consumption) that are not incurred directly by those participating in the activity, but are instead borne by society as a whole. Externalities essentially drive a wedge between private and social costs or benefits, such that decisions based on private costs and benefits (i.e. on market prices) will not be socially optimal, i.e. that the levels of production and pollution will not be socially optimal.

Pollution is an example of a negative externality (an external cost of production or consumption) that is not taken into account by the relevant decision-makers (private costs will be too low relative to social costs). With negative externalities, social costs exceed private costs, such that too much of the activity generating the pollution will be undertaken relative to the socially optimal amount. In this case, market prices will provide incentives for environmentally damaging behaviour. 'Internalising' such externalities is therefore necessary to re-adjust prices in such a way that the negative impacts of pollution will be taken into account by the polluters.

4 Theoretical approach to costing pollution prevention vs. treatment

This section sets out the conceptual framework that was used as guideline in comparing the costs of pollution prevention (e.g. treatment at the point of discharge) and pollution treatment (e.g. treatment at the point of use) from an economic perspective. It was not possible to follow the framework to the letter throughout the study, exceptions were explained in detail. We defined the 'costs' of a particular regime broadly to include the benefits foregone by not implementing the alternative regime (i.e., the benefits of one regime are seen as among the costs of the other regime). It therefore makes more sense to compare the costs of the two regimes (broadly defined) rather than to do a cost-benefit analysis of each regime (i.e., no information will be lost by focusing only on costs, broadly defined, as opposed to both costs and benefits). Furthermore, both regimes involve 'treatment;' the difference lies in the point at which the treatment occurs; i.e. upstream, at the point of discharge (the 'prevention' regime); or downstream, at the point of use (the 'treatment' regime). The cost comparison was based on actual data from one case study area.

The project team assessed and compared the costs of water pollution by means of comparing the social costs (private costs + externalities) of two management regimes within the case study areas. Table 6 outlines the framework for distinguishing between the private and external costs of the two regimes that formed the basis for the cost comparison. Each regime needs to be described in terms of the case study area; while each variable needs to be modeled using real data for the associated pollutant.

Table 6: Framework for estimating costs of prevention versus treatment in two management regimes.

Management Regime 1: Pollution prevention (treatment at the point of discharge)	Management Regime 2: Pollution treatment (treatment at the point of use)
PC1: Private cost to polluter = cost associated with prevention infrastructure (capital outlay, operating and maintenance costs)	PC2: Private cost to user = cost associated with treatment infrastructure (capital outlay, operating and maintenance costs).
EC1: External cost = Any externalities associated with the prevention process	EC2: External cost = externalities associated with pollution between the point of discharge and the point of use; e.g. decrease in public health (loss of labour productivity) and ecosystem service delivery; as well as any externalities associated with the treatment process
Total social cost = PC1 + EC1	Total social cost = PC2 + EC2

Note: PC = private costs; EC = external costs (externalities)

Table 6 suggests that treatment at the point of discharge (regime 1) will incur a private cost (the cost of the water treatment infrastructure) for the polluter or those bearing the cost of such infrastructure, as well as possible externalities associated with the prevention process itself, such as energy use. Treatment at the point of use (regime 2) also incurs a private cost on downstream users (the cost of water treatment infrastructure) and any externalities associated with the process (e.g. energy use); as well as externalities associated with a polluted river, e.g. in the form of health and environmental impacts associated with pollution.

Increased levels of water pollution increase the costs that must be borne by downstream users in order to treat the water to a quality suitable for use. The treatment costs are also dependent on the required target quality of the water. Pollution is considered to be an external cost that is not borne by the polluter, it thus forces a private cost (the cost of water treatment infrastructure and maintenance of such infrastructure) on downstream users. From a private cost perspective, it is therefore highly likely that it will be cheaper to discharge polluted water and do treatment before abstraction; however, this argument excludes external costs. From a social cost perspective, i.e. if external costs are incorporated, it is likely to be the other way around, i.e. prevention is likely to be better than cure. In other words, $\sum$ (PC1, EC1) < $\sum$ (PC2, EC2). The key therefore lies in the quantification of the external costs of not treating pollution at the point of discharge (EC2), i.e. of relying on regime 2 (treatment at point of use). This project attempted to test the hypothesis that prevention is better than cure by quantifying the external costs associated with not preventing pollution at the point of discharge. Note, however, that we cannot claim full inclusiveness regarding the external costs that will be included in the analysis.

Three approaches that can be used to estimate the external cost of pollution are discussed below. We have employed a combination of all three, depending on the pollutant under consideration.

Production function approach: Water is considered to be an important input (production factor) in economic activity. The pollution of water resources reduce the usability of water and hence impacts negatively on the economic productivity of such water. The production function approach holds that water pollution could affect the output, costs and profitability of downstream users through forcing a decrease in the usability of water. Thus, the external cost associated with pollution can be estimated by the value of the change in output, for example, the reduced value of fish caught as a result of river pollution (Bateman *et al*., 2003). It could also be measured in terms of the treatment cost associated with cleaning operations before use.

- Replacement cost approach: The replacement cost approach rests on the fact that negative environmental impacts resulting from pollution could be restored by means of investing in natural capital restoration (Winpenny, 1994; Bateman *et al*., 2003). The cost involved in this restoration process can be used as an estimate of the external cost of the pollution. For example the restoration cost of wetlands could be compared to the water treatment benefits derived from a functional wetland.
- Human capital approach: The human capital approach considers people as economic capital and their earnings as a return on investment. Pollution has negative impacts on the productive capacity of human capital (e.g. a decrease in health leads to a loss of labour productivity). The cost of pollution could be measured in terms of lost working hours (Winpenny, 1994) and therefore a loss of earnings. Another approach that can be used to estimate human health costs is based on the cost of medical treatment that must be incurred as a result of the use of polluted water.

Wastewater Treatment Works (WWTWs) are considered as the main pollution 'prevention' measure (i.e. treatment at the point of discharge as per our definition for the purpose of this study) in the urban/industrial water discharge systems. These Works prevent the pollution water bodies and hence prevents externalities in the system. Pollution 'treatment' (at the point of use) refers to water treatment works (water purification works, i.e. WPWs) which present only one of the costs associated with pollution (i.e. WPW would have been unnecessary if the environment was in a pristine and unpolluted state, however because of pollution, certain cost needs to be incurred before the water could be used.

The intended comparison of the costs of pollution prevention vs. treatment was based on several factors that, together, defined the costs associated with prevention as opposed to treatment of polluted water:

- The impact of industry waste on the successful operation of municipal WWTW
- The cost of WWTW operations in terms of treatment and upgrading of facilities to treat all waste to the required levels
- The impact of high density residential areas with limited or non-functional waste water treatment services
- The impact of agricultural practices on pollution and the impact of polluted water on agriculture (irrigation and crop quality)
- External costs associated with the use of unpurified water contaminated with unacceptably high levels of microbial pollution and algal (and possibly other)

toxins; particularly by recreational users and areas without formal service delivery or where the available water is of poor quality

- The cost of water purification works
- The impact of different forms of pollution on water purification activities
- The costs associated with poorly treated water, including de-sludging of bulk water reservoirs and possible upgrading/expansion of this treatment infrastructure

The following three sub-sections describe the private and external costs as per Table 6 as they relate to our case study area. External costs associated with the 'prevention' and 'treatment' processes (e.g. energy use) will be ignored, based on the simplifying assumption that both regimes are equally subject to these types of externalities. It is therefore assumed that there are no external costs associated with the 'prevention' regime; although there are external costs associated with the 'treatment' regime in the form of pollution impacts between the point of discharge and the point of use.

4.1 Private costs associated with pollution prevention (PC1)

Pollution prevention in an industrially developed area, such as the City of Johannesburg and Tshwane municipalities, can be directly linked to the formal WWTWs in these areas. The Water Services Act (RSA, 1997) requires that all industrial wastewater from municipal areas must be discharged into the sewer systems. Industries pay a levy to the Municipalities based on the quality and quantity of this effluent. Fines may also be instituted against any unlawful release into the sewer system of effluent streams by industry. The level of treatment of their effluent by individual industries should be carefully co-ordinated with the relevant municipalities. Certain waste streams provide chemical oxygen demand (COD), necessary for nutrient removal in the WWTWs. Other wastes (e.g. oils, pharmaceuticals, and medical) are more toxic or damaging to waste water treatment processes, and should be pre-treated by the industry in question before release into the system. Co-operation between industry and the local municipal authorities regarding accidental releases of toxic substances or high volume waste loads also benefits the successful operation of WWTWs. The system in place for dealing with industrial effluent streams by municipal and private WWTWs represent the most important pollution prevention barrier in the urban/industrial system. The largest contribution regarding the costs of pollution prevention can therefore be obtained by establishing the costs associated with WWTWs and by estimating the capital and other costs required to keep these treatment works operating. Well operated WWTW should deliver effluent quality that meets the required standards for discharge. Any additional efforts by industry to pre-treat their effluent streams before discharge into the municipal sewerage system, will benefit the WWTWs and reduce the costs of treatment to the municipality. The costs associated with providing appropriate and adequate sanitation systems to high density settlements and the necessary educational efforts to ensure the correct usage thereof by the local population, should also be included.

In our case, the private costs associated with pollution prevention primarily involves the costs incurred in operating the WWTWs to ensure that final treated effluent quality meets the required standards. This will include upgrading/expansion of infrastructure and modernisation of technology. Any costs incurred by industry for pre-treatment of their effluent, should be considered as an additional pollution prevention cost. The cost of prevention measures aimed at reduction of diffuse pollution will also be factored into the equation. Another important cost factor will be the estimated costs of repairing and expanding the waste treatment infrastructure in and to high density residential areas to reduce diffuse pollution emanating from such areas.

4.2 Private costs associated with pollution treatment (PC2)

The private costs associated with pollution treatment include the direct cost impacts of pollution (when prevention is lacking or inadequate) on water treatment. The impact of pollution on water purification for potable use can be obtained by studying the costs of water purification at different downstream sites in the river system selected for the study. The progressive increase in pollution in the system can be traced using analytical data regarding the water quality in the system. This information can then be linked to the processes and costs of the various WPWs in the same river system. By carefully dissecting the costs of water purification at the different points in the system and linking this information to the type of pollution impacting on the different treatment processes (process costs or expansion/upgrading of infrastructure), it will be possible to determine the direct impact of specific pollutants on the costs of water treatment in the selected river system. It should be noted that microbial contamination and salinity levels of the source water has little impact on the costs of water purification at this point in time. Salinity is not currently a problem in the selected areas of study (but may impact on irrigation of sensitive crops downstream). The efficiency of the final chlorination stage at the WPW is however affected by the overall water chemistry e.g. high levels of ammonia reduce the efficiency of the chlorination.

Increased costs to agriculture include the upgrading of irrigation equipment and systems and/or water treatment at abstraction points due to increased levels of pollution. Finally, costs associated with sediments and salinity may include loss of storage capacity, corrosion and/or scaling of infrastructure, including pipes, pumps etc. The cost of treatment of sedimentation will equate to dredging and or flushing of dams and lakes as well as creating additional storage capacity through raising of dam walls or building of new dams.

4.3 External costs associated with pollution treatment (EC2)

The external costs associated with the treatment regime (regime 2) include the costs that arise as a result of not undertaking prevention (regime 1); i.e. not treating at the point of discharge; in other words, the impacts of pollution that remains in the system between the point of discharge and the point of treatment. For example, it seems logical to assume that a pristine environment due to the good quality of water in a river system will increase the value of the surrounding farm land and/or real estate. However, if water is not treated at the point of discharge, this good quality of water (and the associated benefits) will not be lost. These foregone benefits can be seen as a cost associated with the failure to prevent water pollution (i.e. an external cost associated with regime 2). Similarly, microbial pollution and salinity can have severe effects on the viability of irrigated crop production and livestock farming. The potential loss of income to agriculture due to microbial pollution and salinity must also be estimated and included in the model; as must any costs associated with human health impacts. These types of costs can, for example, be estimated based on:

- the increase in the value of real estate or farmlands linked to a pristine river system or dam (when pollution is prevented successfully),
- the benefits associated with improved recreational use of water, and
- the costs incurred due to illnesses associated with waterborne diseases.

The private and external costs associated with the two management regimes can be summarised as in Table 7.

Table 7: Private and external costs associated with prevention versus treatment in the case study area

Cost of pollution prevention = ∑ (…)	**Cost of pollution treatment** = ∑ (…)
Capital outlay, maintenance and operating costs associated with WWTWs (PC1)	Capital outlay, maintenance and operating costs associated with water purification works (WPWs) (PC2)
Capital outlay, maintenance and operating costs associated with industrial treatment (PC1)	Capital outlay, maintenance and operating costs associated with dealing with sediment (PC2)
Capital outlay, maintenance and operating costs associated with prevention of agricultural run-off (PC1)	Health-related costs associated with untreated or poorly treated water (e.g. decrease in public health and associated loss of labour productivity; and increased medical costs) (EC2)
Capital outlay, maintenance and operating costs associated with ensuring high level waste water treatment in high density residential areas (PC1)	Decrease in agricultural income and land value due to the decline of the water quality of a river if pollution (e.g. microbial pollution of irrigated crops and livestock) is not treated at the point of discharge (EC2)
	Decreased value of river systems and loss of ecosystem service delivery; for example, loss of recreational and real estate value and the value of a healthy ecosystem (EC2)
Total social cost = PC1 + EC1	Total social cost = PC2 + EC2

5 Study areas

The framework for a cost comparison as outlined in the previous section provides a reasonable approximation of the difference in costs between pollution prevention and pollution treatment. However, obtaining the required information at an acceptable level of detail and confidence proved to be a challenge. For this reason, it was decided to focus the research at even smaller, limited river systems with the potential to expand this framework to other areas in future if information becomes available.

An in-depth analysis of a specific system within a water management area was required. The delimitation and choice of a specific study area was based on the following criteria:

- The number of municipalities in the service were to be kept at a minimum for logistical reasons;
- Availability of relevant and current analytical data for the area;
- Areas presenting different levels of pollution;
- Presence of water purification works (potable water treatment facilities) (WPW);
- Presence of wastewater treatment works (sewage works) (WWTW);
- Representation of a selection of water users (industries, irrigated agriculture; and formal and informal housing with varying levels of water supply, sanitation and waste removal services.

Portions of two water management areas (WMAs), the Crocodile (West) and Marico WMA upstream of the Hartbeespoort Dam and the Olifants WMA were selected, mainly due to the availability of data and the strategic importance of these catchments to the local and national economy. Data from these two study areas supplemented with data at a national level were used to determine the cost impacts of pollution on water users.

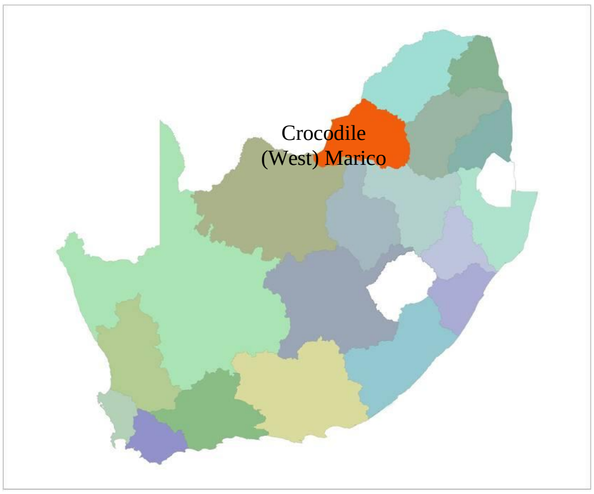

Figure 2: Geographic location of the Crocodile (West) Marico WMA (courtesy of Wilma Strydom)

5.1 Crocodile River in the Crocodile (West) and Marico Water Management Area

The Crocodile River catchment in the Crocodile (West) and Marico water management area, spans portions of the Gauteng, North West and Limpopo provinces. It is a tributary of the Limpopo River which discharges into the Indian Ocean in Mozambique. The Pienaars, Apies, Moretele, Hennops, Jukskei, Magalies and Elands Rivers are the major tributaries of the Crocodile River. The Upper portion of the Crocodile (West) river catchment, south east of Hartbeespoort Dam is located in Gauteng Province. The north and north-east corners lie in the Limpopo Province and the central and western sections in lie in North West province (DWAF 2004c).

The total area of the Crocodile River catchment is 29 400 km^2. There are 9 major storage dams in the catchment (DWAF, 2004c). The Rietvlei and Hartbeespoort dams form part of the bulk water supply infrastructure in the upper portion of the area. Large quantities of water are transferred into the catchment via Rand Water's bulk distribution system from the Upper Vaal WMA to the northern parts of Johannesburg, the greater Pretoria area and Rustenburg. Almost 520 million m^3 of water were transferred in the year 2000 (DWAF, 2004c).

The population of the Crocodile River catchment reside mainly in the urban areas of City of Johannesburg and City of Tshwane metropolitan municipalities. The Upper Crocodile sub-area hosts a population of around 2.2 million. Another 2.2 million stay in the Apies-Pienaars sub-area and only about 150 000 people in the Lower Crocodile portion of the catchment (DWAF, 2004c). A large portion of the population is under 20 years of age, with an unemployment rate of between 30 to 40%. Many of these people stay in the informal settlements around Johannesburg, Pretoria and Rustenburg (DWAF, 2004c).

The urban areas of northern Johannesburg, Midrand and the City of Tshwane Metropolitan municipality dominate the land-use in the south-eastern portion of the catchment. Urban areas cover an area of 665 km^2 and activities in these areas make up a significant portion of the economic activity (i.e. service and government sectors, manufacturing and industry). Together with mining activities, close on a quarter of the country's GDP is generated in this area (DWAF, 2004c).

Smallholdings and commercial agriculture take place in the area to the north west of Johannesburg, but south of the Magaliesberg Northern Range. The area between Rustenburg and Brits on the northern side of the Magaliesberg range is known for citrus farming activities whereas irrigated cash crop farming takes place below the Hartbeespoort Dam and Brits. Irrigation farming occurs along the mainstream of the Crocodile River, the most significant areas being just south and north of Thabazimbi. About 650 km^2 of irrigation has been recorded (DWAF, 2004c).

The remainder of the area is used for dry land farming (limited), cattle grazing and game farming.

There are three relatively small thermal power stations in the area:

- Kelvin power station situated in Kempton Park. This station is supplied with approximately 12.8 million m^3/annum treated effluent from the Johannesburg Northern waste water treatment works.
- Tshwane power station situated in Pretoria West, next to Iscor Iron and Steelworks. It receives 6 million m^3/annum of treated effluent from the Daspoort waste water treatment works.
- Rooiwal power station north of Pretoria. Treated effluent at 7.7 million m^3/annum, is supplied from the Rooiwal waste water treatment works adjacent to the power station.

The primary minerals that are mined in the catchment include: platinum and platina group minerals, gold, chrome, manganese, iron ore, diamonds, mineral sands, vanadium, limestone and andalusite. Granite is also mined in this area (DWAF, 2004c). The mines draw part of their water supply from the Vaal River System while the remainder is obtained from groundwater or local dams.

The majority of the gold mines situated in the upper portion of the Crocodile River catchment, are mined out and have been closed. Acid mine drainage from defunct and flooded underground mine workings on the West Rand started decanting in 2002. Decant has subsequently been reported at various mine shafts and diffuse surface seeps in the area (Oelofse *et al.,* 2007). This AMD is flowing into the upper Crocodile sub-area.

Industries are primarily situated in the peripheral industrial zones in and around Johannesburg and Pretoria. Industrial water supply in these areas is through bulk supply from local authorities at drinking water quality. The industrial effluent is discharged into the sewer systems of the local authorities. The quality and quantities of industrial effluent received into the sewer systems are regulated by the municipalities through by-laws and tariffs systems. The municipality is responsible for treating the effluent to appropriate discharge standards at the waste water treatment works.

5.1.1 Water Quality in the catchment

The water quality in this area deteriorates progressively downstream, with increased levels of suspended solids, nutrients and microbial contamination.

The hypertrophic Rietvlei Dam receives discharges from the Hartbeesfontein (45 Mℓ/day) waste water treatment works (WWTW), 40 km upstream of the dam. Wastewater from Thembisa, Ivory Park, Rabie Ridge and Olifantsfontein are treated, resulting in 14 million m^3/annum of effluent being discharged into the Swartspruit, upstream of the Rietvlei Dam, adding salt and nutrients to the system (Oberholster *et al.*, 2008).

The Hennops River catchment, adjacent to the Rietvlei Dam drainage area, has a water quality problem due to uncontrolled urban development along the Kaalspruit. Informal brick making activities along the Kaalspruit represents a significant source of suspended solids, leading to excessive sedimentation of the Centurion Lake.

A number of factors, including informal settlements without access to sanitation, sewage spills from poorly maintained or overloaded sewage networks (Sandton and Alexandria), and industrial and agricultural pollutants have led to high levels of nutrients in the rivers feeding the Hartbeespoort Dam. The build-up of these nutrients in the Hartbeespoort Dam is well known and the subject of many research reports.

5.1.2 Study area 1

The selected study area, representing an urban/industrial context, is therefore limited to the upper portion of the Crocodile (West) catchment up to and including the water purification works (WPW) near the town of Brits downstream of Hartbeespoort Dam A schematic illustration of the study area is provided in Figure 3. The sources of pollution in this study area are well defined and the WWTW identified by its corresponding number in Table 9.

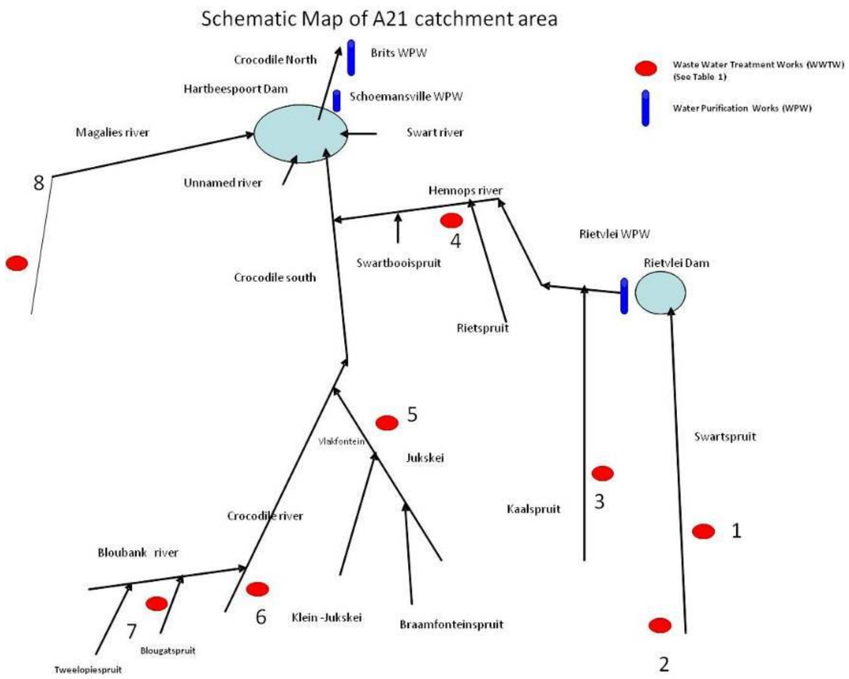

Figure 3: A Schematic diagram of the Crocodile (West) Marico study area

A study by Oberholster *et al.* (2008), done prior to the upgrading of the Hartbeesfontein WWTW, indicates that the quality of the water flowing into the Rietvlei Dam was being seriously affected by the poorly operated Hartbeesfontein WWTW. The nutrient pollution and resulting algal blooms in the Rietvlei Dam sparked the upgrading of this WWTW (R70 million) to avoid future increase of the nutrient (especially phosphorous) content in the dam. The high nutrient load and resulting eutrophication of the Rietvlei Dam had a significant impact on treatment costs and technology deployed by the Rietvlei WPW (Oberholster *et al.*, 2008).

The WPW has to deal with the effects of algal blooms and can therefore provide important insights into the effect of eutrophication on water treatment costs. The effects of suspended solids, salinity and microbial contamination are negligible regarding the current treatment processes required at the Rietvlei WPW for the production of potable water.

The Kaalspruit is the point of release of treated effluent from the Olifantsfontein WWTW (105-110 Mℓ/day) that treats wastewater from Thembisa and Midrand. Dredging and desludging of the Centurion Lake have a direct cost impact on the City of Tshwane metropolitan municipality.

By extending the study area to include the Hartbeespoort Dam WPW as well as that of Brits in the Madibeng municipal area, the further influence of additional pollution stemming from the Hartbeespoort Dam itself can be investigated. However, the sources of pollution impacting on the quality of the Hartbeespoort Dam are not the immediate concern of this investigation; instead, the focus is on the effects of this pollution on the costs of downstream water treatment systems.

5.2 Olifants River in the Olifants Water Management Area

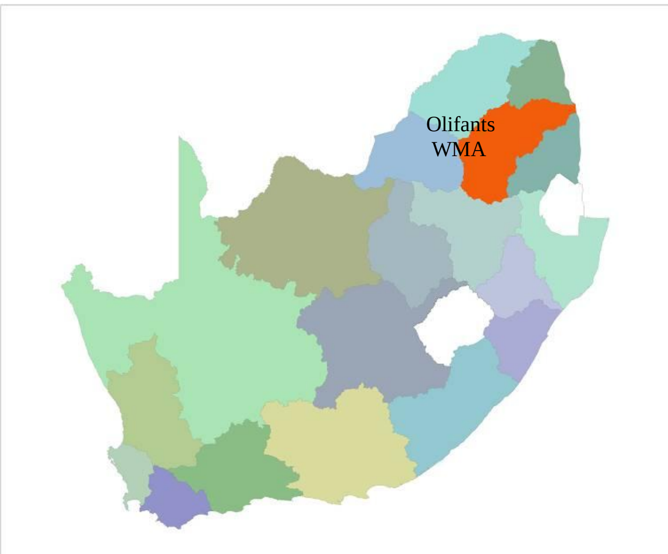

Figure 4: **Geographic location of the Olifants WMA** (courtesy of Wilma Strydom)

The Olifants River in the Olifants WMA (Figure 4) rises east of Johannesburg and flows in a north north-eastern direction into Mozambique, through the Mpumalanga and Limpopo provinces. The Olifants River is one of the largest rivers flowing through the Kruger National Park, which is an important tourist attraction in South Africa. The Olifants River basin is an important sub-catchment of the Limpopo River basin (Basson *et al*., 1997).
The upper Olifants catchment area is approximately 11 464 km^2 and stretches across the city of Witbank and town of Middelburg (Hobbs *et al*., 2008). The Bronkhorstspruit Dam, Witbank Dam and Middelburg Dam as well as the Loskop Dam fall within this catchment (Basson *et al*., 1997) where coal mining significantly effects the hydrological cycle. The middle Olifants sub-catchment is dominated by agriculture while the Steelpoort sub-catchment is dominated by mining and agriculture. The lower Olifants sub-catchment is dominated by tourism and mining.

Several major tributaries including the Wilge, Moses, Elands and Klein Olifants rivers, as well as the Steelpoort and Blyde rivers flow into the Olifants River. In the lower reaches of the Olifants River catchment, the Letaba River joins the Olifants River inside the Kruger National Park, a short distance upstream of the Mozambique border (Basson *et al*., 1997).

The Olifants River supports a population of approximately 3.4 million (DWAF 2002b). A relatively small proportion of the basin's population lives in urban areas with modern domestic water supply and sanitation systems. The largest proportion lives in rural areas or small towns with rudimentary or no formal domestic water supply system and no or inadequate sanitation systems. Population densities across the catchment are illustrated in Figure 5Figure 5.

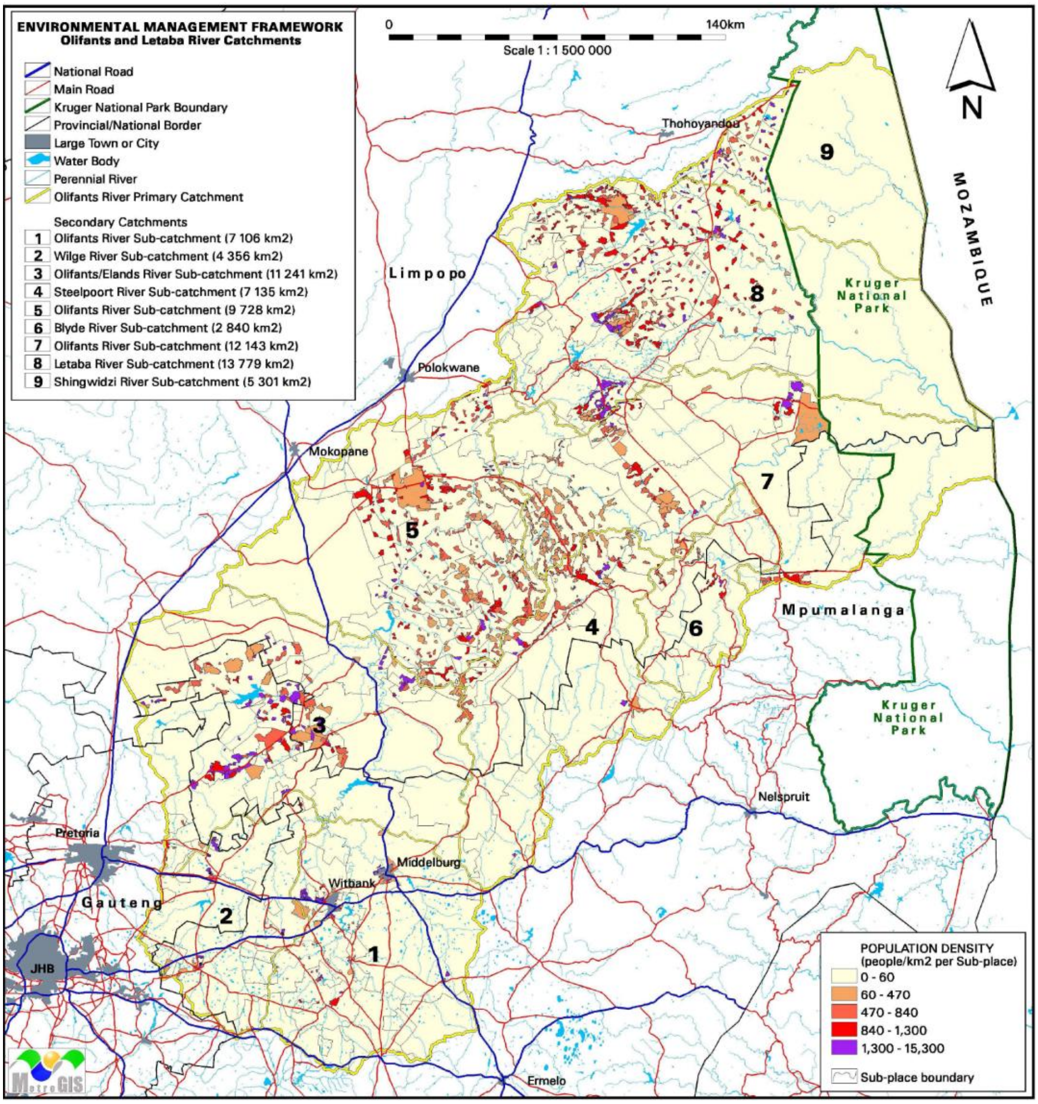

Figure 5: Population densities in the study area (EMF, 2009)

Agricultural activity in the upper and middle reaches of the Olifants catchment area consists mainly of large scale irrigation. Agricultural activities are intensified along the middle and lower parts of the catchment (Grobler *et al*., 1994). The irrigation scheme of the Loskop dam consists of seven small balance dams and a total of 480 km of canals (Ferreira, 2009). The scheme currently serves 16 136 hectares or 630 registered allocations with an average enlistment of 25 hectares at 7 700 m^3/hectare/annum. The enlistments include supplies to the

Hereford and Olifants River Irrigation Boards, as well as the Groblersdal and Marble Hall local municipalities. The main commercial crops under irrigation in the area include wheat, maize, vegetables, tobacco, peanuts, cotton and citrus.

Six of the world's largest coal-fired power stations are located in the upper Olifants catchment area (DWAF, 2004a). The region supports 48% of the country's total power generating capacity (Tshwete *et al.*, 2006) for export and domestic consumption (Hobbs *et al.*, 2008).

The upper reaches of the catchment is dominated by mining (Grobler *et al.*, 1994). Opencast coal mining operations in the region were already underway in the early 1970s, and disturbance of the land is massive compared to the earlier underground workings (Cochrane, 2002). Increased levels of metals, deriving from mining in the upper Olifants River, have been reported in the Loskop Dam (Engelbrecht, 1992).

The management of the mine discharges and post-closure decants from operating and defunct coalmines in the region for potential reuse is being assessed by the mining companies (DWAF, 2004a; Hobbs *et al.*, 2008). Although coal mining activities are expected to decline over the next 20 years (DWAF, 2004a), the decanting of AMD in this area is expected to continue into the future for decades to come.

Industrial activities in the catchment are closely associated with Witbank and Middelburg in the upper Olifants catchment, Burgersfort in the Steelpoort Valley and Phalaborwa in the lower Olifants sub-catchment. Metallurgical industries dominate the industrial activities in the Olifants WMA (DWAF, 2004a).

5.2.1 Water quality in the Olifants River Catchment

There are a number of water quality issues in the Olifants River catchment area as indicated in Figure 6 and discussed in Table 8 (EMF, 2009). The colour coding used in Table 8 corresponds with the colour coding indicated in Figure 6.

Table 8: Water quality in the river reaches of the Olifants River Catchment (EMF, 2009)

River reach by segment number	Description	Comments
Upper Olifants		
Olifants 1-8	Olifants river from its source to the confluence of the Steenkoolspruit	The Upper reaches of the Olifants River are relatively undisturbed with dry land agriculture being the main land-use and some coal mining at the bottom end of the reach.
Olifants 9-13	Olifants River from the Steenkoolspruit confluence to the inflow into the Witbank Dam	This reach is highly impacted by coal mining and power generation activities as well as poor water quality entering the reach via the Steenskoolspruit.
Olifants 14-27	Olifants River downstream of the Witbank Dam to the Klipspruit Confluence	This river reach is highly impacted by water from the Spookspruit (due to coal mining activities) and the Klein Olifants River. There are no DWAF water quality monitoring points in this reach which can be used to determine the present ecological state.
Olifants 28	Olifants River from the Klipspruit confluence to the Wilge River Confluence	This river reach is negatively impacted by the poor water quality in the Klipspruit (due to old coal mining activities). There are no routine DWAF monitoring stations in this reach which can be used to determine the present ecological state

River reach by segment number	Description	Comments
Olifants 29-37	Olifants from the Wilge River confluence to the inflow into the Loskop Dam	This reach is positively impacted by good water quality in the Wilge River. There are no routine DWAF monitoring stations in this reach which could be used to assess the present ecological state.
Klein Olifants 1-4	Klein Olifants upstream of Middelburg Dam	The Klein Olifants river is highly affected by coal mining and power generation activities
Klein Olifants 5-12	Klein Olifants from downstream of Middelburg Dam to the confluence with the Olifants River	There are no routine DWAF monitoring stations in this reach and the weir downstream of the Middelburg Dam used.
Wilge 1-6	Bronkhorstspruit from Bronkhorstspruit Dam to Premier Mine Dam	This reach is relatively un-impacted and agriculture is the main land-use activity with minor sewage discharges at Bronkhorstspruit.
Wilge 7-20	Wilge River from the Premier Dam to the confluence with the Olifants River	This reach is in good condition with agriculture as the main land-use activity.
Middle Olifants		
Olifants 39-57	Olifants River from Loskop Dam to Flag Boshielo Dam	This reach is highly impacted by irrigation activities at the Loskop irrigation Scheme as well as the Moses and Elands Rivers which also receives irrigation return flows.
Olifants 58-84	Olifants River from Flag Boshielo Dam to segment 84 downstream of Mohlapitse confluence	This reach of the Olifants River has the worst water quality. It is probably the result of irrigation return flows, poor land-use practices adding substantial suspended sediment loads and evaporation losses concentrating salts in the river.
Elands River 1-15	Elands River from its source to the inflow into the Renosterkop Dam	The upper Elands is in a moderately good condition with irrigation return flows below the Rust de Winter Dam adding salts to the system
Elands River 16-27	Elands River from downstream of the Rhenosterkop Dam to the confluence of with the Olifants River	The lower Elands is highly impacted by irrigation return flows from the Loskop irrigation scheme (which drains to the Olifants and Elands Rivers and semi-urban development at Siyabuswa.
Lower Olifants		
Olifants 85-99	Olifants River from segment 84 to Blyde River confluence	This reach of the Olifants River is mostly affected by water quality in the Steelpoort River
Olifants 100-110	Olifants river from the Blyde River Confluence to the Selati Confluence	The Blyde River improves the quality of the Olifants River in this reach, Especially during low flow months
Olifants 111-132	Olifants river from the Selati confluence to downstream border of the Kruger National Park	This reach of the Olifants River is negatively impacted by poor water quality in the Selati River as a result of the mining activities at Phalaborwa.
Steelpoort 1-8	The whole of the Steelpoort River	Water quality in the Steelpoort River is affected by mining activities in the Steelpoort area and irrigation activities in the Spekboom catchment in the Burgersfort area.
Blyde 1-8	The whole of the Blyde River	Water quality in the Blyde River is relatively good and has a positive impact on the Olifants River
Selati 1-9	Upper Selati River from its source to Selati ranch	Water quality in the upper Selati river is generally good. Irrigation is a major water use in this part of the catchment which has a moderate impact on the water quality.
Selati 10-18	Lower Selati river from Selati ranch to the confluence with the Olifants River	Water quality in the bottom end of the Selati river is very poor as a result of water discharges from slimes dams and domestic effluent discharges in the Phalaborwa area.

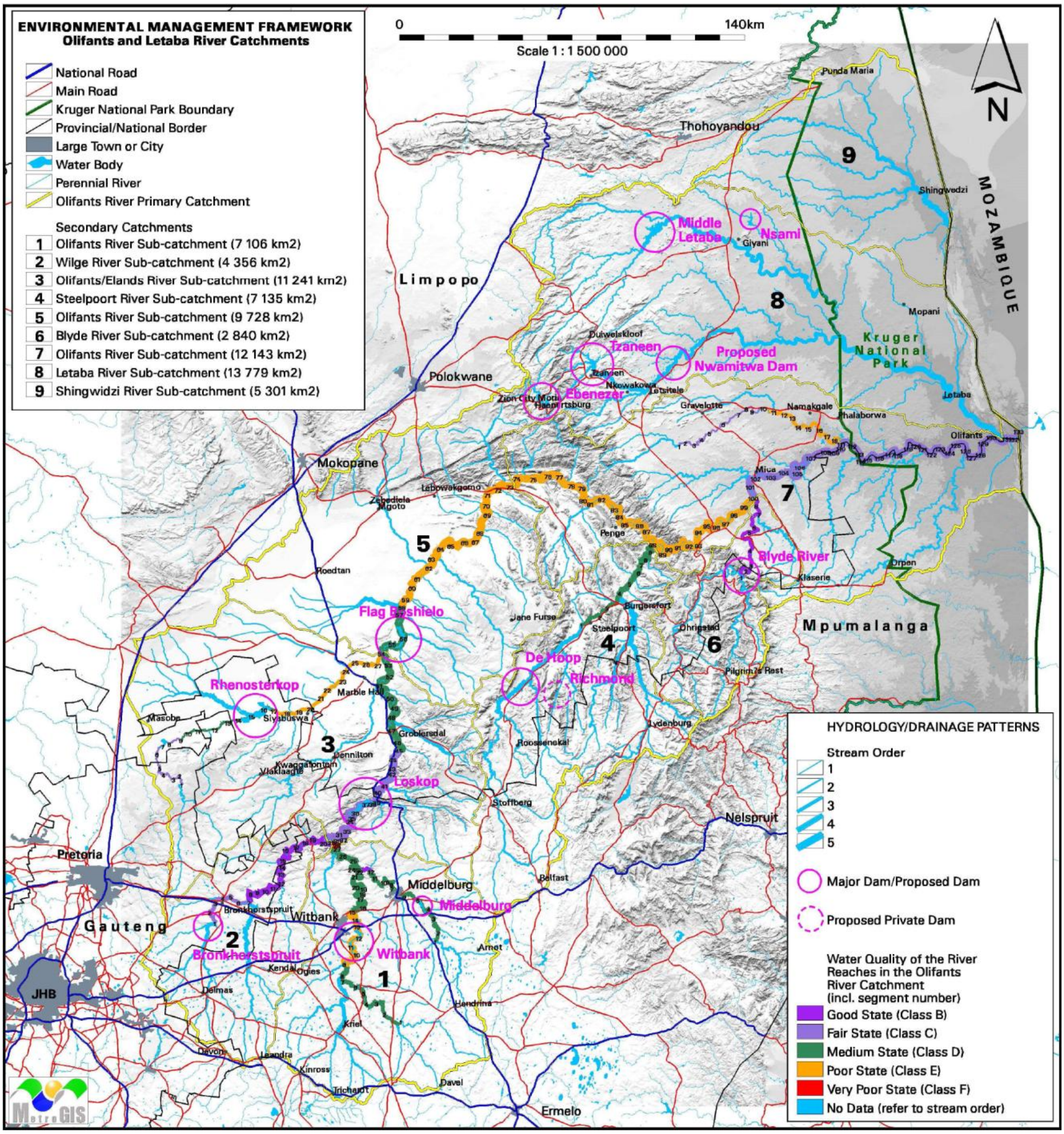

Figure 6: Indication of the overall water quality in the Olifants River (EMF, 2009)

5.2.2 Study area 2

The selected study area, representing an agricultural/rural context, stretches from east to west with Delmas, Bethal and Hendrina in the south, northwards past the towns of Bronkhorstspruit, Witbank and Middelburg, the Loskop Dam and further north past Groblersdal, Marble Hall to just down-stream of the Flag Boshielo Dam. A schematic overview of the area south of the Loskop Dam is provided in Figure 7 while the area north of Loskop Dam to just downstream of the Flag Boshielo Dam wall is dealt with in Figure 8. According to available information, the system meets the requirements of most of the selection criteria mentioned. The main sources of pollution are fairly well defined and identifiable.

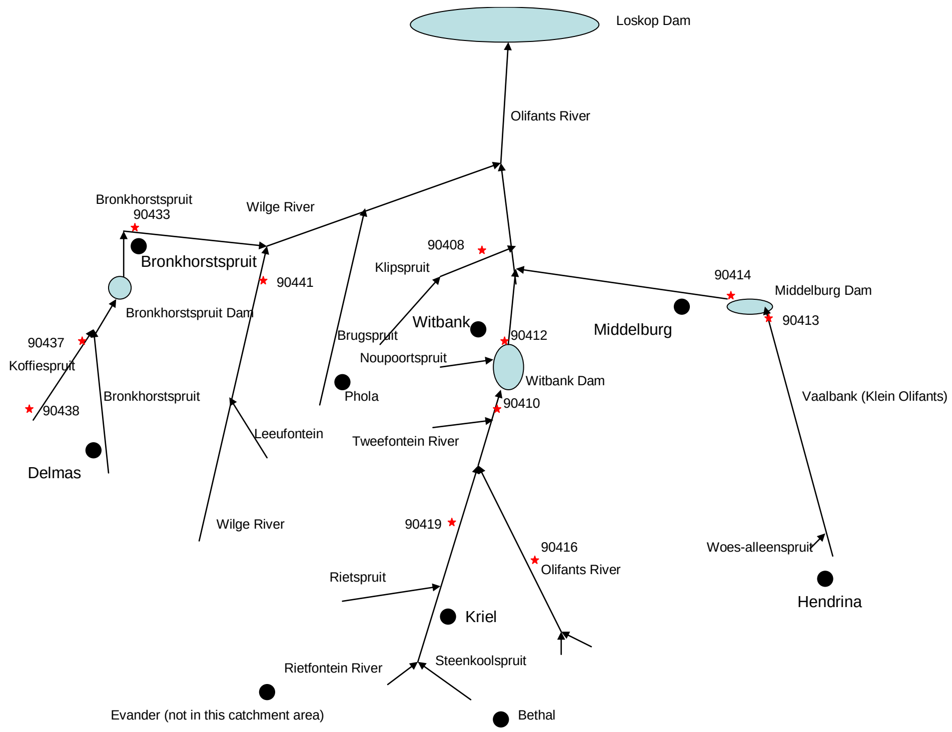

Figure 7: A schematic drawing of the rivers and DWA sampling points (indicated with *) in the study area south of the Loskop Dam

The area upstream of the Loskop Dam is impacted mostly by mining and power generation. The land use between the Loskop Dam and Flag Boshielo Dam is mainly irrigated agriculture. At the Loskop Dam wall (Figure 8) two irrigation channels, one on the left and one on the right hand side of the Olifants River, leave the dam to supply the extensive irrigation scheme in the Groblersdal area. This irrigation scheme is largely parallel to the Olifants River and ends south of Marble Hall. The Olifants River is joined by the Selons River from the east, just downstream of the Loskop Dam and by the Bloed River from the east, just north and downstream of Groblersdal. Both these rivers flow from natural catchment areas and carry relatively unpolluted water into the Olifants River. The impacts of salinity, associated with mining and power generation, on agriculture can be estimated in the irrigation area of the Loskop Dam irrigation scheme downstream of Loskop Dam and up to Flag Boshielo Dam

The water quality in the section of the study area downstream of the Renosterkop Dam and is heavily impacted by irrigation return flows from the Loskop irrigation scheme and the semi-urban development at Siyabuswa.

The WWTW of Groblersdal discharges into the Olifants River while the WPW abstracts its water from the Olifants on the west bank of the river and upstream of the treated sewage discharge point. The Marble Hall WWTW also discharges to the Olifants or Elands rivers and is situated in the town itself. Very little is known of the service delivery for water and sanitation in the townships on the Moses and Olifants rivers although local residents report that water services in the area comprise mainly pit latrines and communal taps. A large portion of the population in the study area lives in rural areas with poor access to water and sanitation services. The situation in this study area will enable the project team to estimate the cost of water pollution on human health.

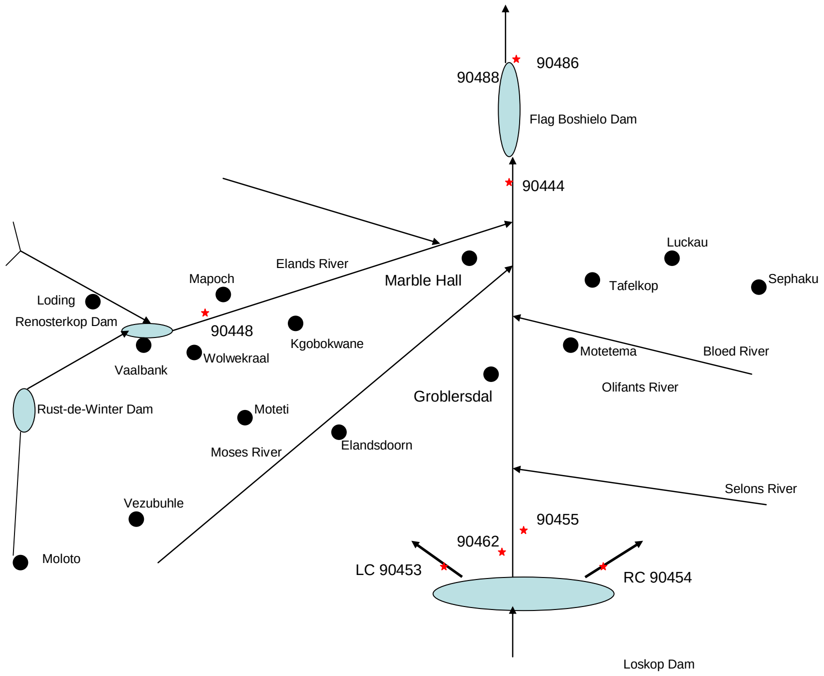

Figure 8: A schematic drawing of the rivers and DWA sampling points in the study area between Loskop Dam and Flag Boshielo Dam

The fairly recent raising of the dam wall at the Flag Boshielo Dam and the current construction of the De Hoop Dam in the Steelpoort area provided the project team with the required information to estimate the cost of loss of storage capacity as a result of sedimentation in dams.

6 Cost of Eutrophication

6.1 Eutrophication in study area 1

Both of the large dams included in study area 1, Rietvlei and Hartbeespoort dams, are on the list of top ten impoundments in need of nutrient management (DWAF, 2002a). Water quality monitoring data suggest that a large portion of the phosphate loads entering the Hartbeespoort Dam is trapped in the dam (Figure 9). It is therefore concerning to note the increasing trend (Figure 10) in phosphate loads in the Crocodile River upstream of the Hartbeespoort Dam.

Wastewater treatment plants are major point sources of nutrient pollution, while diffuse sources include stormwater run-off and agricultural return flows. Agrochemicals (fertilizers and pesticides) have a negative impact in the study area but the extent of this problem is not well known in the catchment.

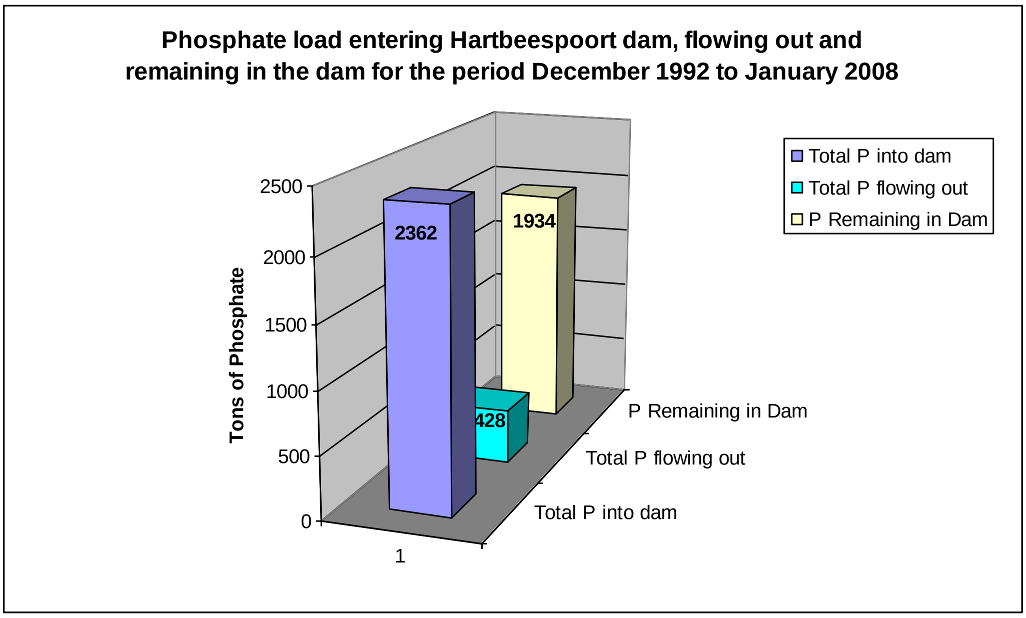

Figure 9: Phosphate loads in the Hartbeespoort Dam (Roux and Oelofse, 2010).

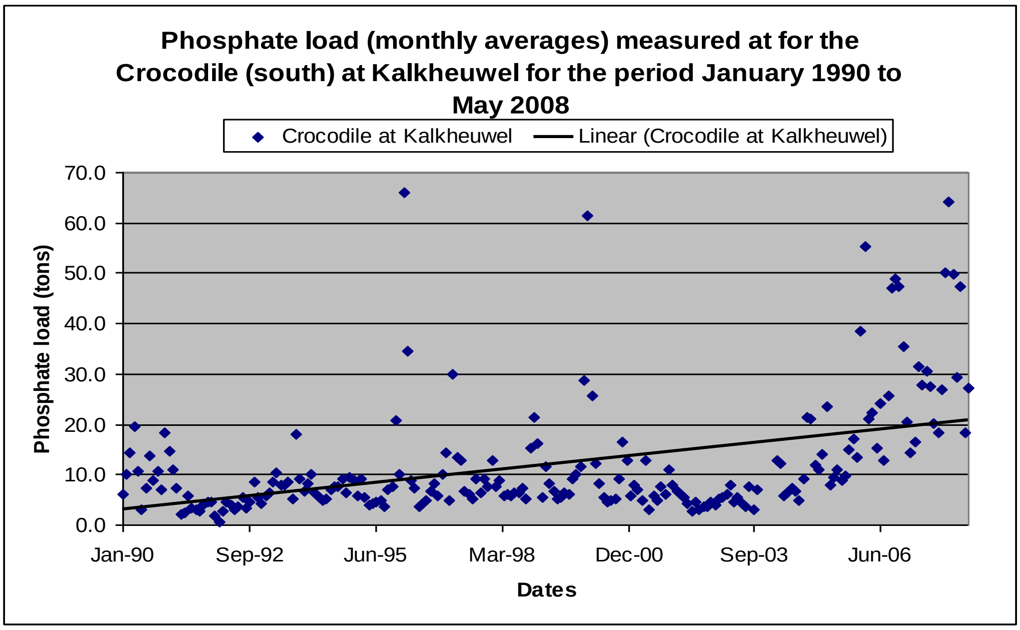

Figure 10: Trends in phosphate load at Kalkheuwel, upstream of the Hartbeespoort Dam (Roux and Oelofse, 2010)

6.1.1 Point source discharges contributing to the nutrient load

There are currently nine wastewater treatment plants discharging into the Crocodile River and its tributaries upstream of the Hartbeespoort Dam. Treatment standards at certain of these sewage treatment works have however been lowered due to various reasons (DWAF, 2004c). The lowering of treatment standards at sewage treatment works however only adds to the pollution problem in the catchment.

Details, including treatment capacity and average flow, at eight of the WWTW in study area 1 are provided in Table 9.

Table 9: WWTWs in the study area

No	WWTW	Discharge River	Capacity (Mℓ/day)	Average flow (Mℓ/day)	Operating costs R/Mℓ	Planned expansion of facilities	Replacement value/ Capital value in R
1	Hartbees-fontein (6)	Swart-spruit	45	50	1 500 (an average price for all ERWAT WWTWs)	New 120 Mℓ/day WWTW on the Swartspruit. Phase one (50 Mℓ/day) to be completed in 2013 @ R260m	315m at R7m/Mℓ.day
2	Esther Park (6)	Swart-spruit	0.4	0.4	See Hartbees-fontein		
3	Olifants-fontein (6)	Kaalspruit	105	70	See Hartbees-fontein		735m at R7m/Mℓ.day
4	Sunderland Ridge (7)	Hennops River	65	58	794.1	Increase capacity to 95 Mℓ/day by 2010-2013 @ R300m; New 50 Mℓ/day WWTW near Skurweberg on Hennops River to be completed in 2016 @ R260m	585m at R8m/Mℓ.day
5	Northern Works (8)	Jukskei	450	380		Phase two to be completed in 2013 with phase 3, (an additional 50 Mℓ/day) planned for 2025	2 700m at R6m/Mℓ.day
6	Driefontein (8)	Crocodile	35	35		Expansion of additional 25 Mℓ/day @ R150m	210m at R6m/Mℓ.day
7	Percy Stewart (9)	Blougat Spruit	15	18		Increasing the capacity to total 25 Mℓ/day by 2012 at a cost of R94.3m	139.5m at R9.3m/Mℓ.day
8	Magalies	n/a	n/a	n/a	n/a		

Note: The capital (current replacement) value is supplied according to the information supplied by the owner/operators of the different WWTWs. Various factors including location, specific design and size determine that the different WWTWs are valued at different capital amounts/Mℓ.day (Roux and Oelofse, 2010).

6.1.2 Diffuse source discharges contributing to the nutrient load

Informal settlements with onsite sanitation and poor grey water management are also major diffuse sources of nutrient pollution. The situation in the study area is summarized Table 10 based on the 2007 Community Survey (Stats SA, 2007) data.

Table 10Table 10: Overview of water, sanitation and informal housing in study area 1(% households) (Stats SA, 2007)

City	Informal Housing	Main water supply source	Sanitation
Johannesburg	In Backyard 8.4% Squatter camp 10.4%	Indoor pipe 70.8% Yard tap 20.8% Communal Tap 6.7% Stream 0.1% Water vendor 0.9% Borehole 0.2% Other 0.5%	Sewerage system 86.8% VIP 3.2% Pit without ventilation 0.8% Bucket 1.5% Dry toilet 1.6% Chemical toilet 2.1% Septic tank 2.7% None 1.2%
Tshwane	In Backyard 7.1 % Squatter camp 19.7%	Indoor pipe 62.5% Yard tap 18.2% Communal Tap 16.4% Water vendor 1.2% Borehole 0.6% Spring 0.1% Other 0.8%	Sewerage system 71.3% VIP 20% Pit without ventilation 0.2% Bucket 0.8% Dry toilet 3.0% Chemical toilet 1.5% Septic tank 1.8% None 1.4%
Rustenburg	In Backyard 21.0% Squatter camp 16.3%	Indoor pipe 37.4% Yard tap 40.7% Communal Tap 8.5% Water vendor 9.8% Borehole 1.2% Spring 0.1% Other 2.3%	Sewerage system 51.5% VIP 31.0% Pit without ventilation 0.1% Dry toilet 1.2% Chemical toilet 6.1% Septic tank 4.5% None 5.6%

6.2 Eutrophication in study area 2

Phosphate loads and total nitrogen loads (combined value for ammonia and nitrates as N) were investigated to gain insight into potential eutrophication of the Olifants River and the dams in study area 2. The accuracy of the data suffers from a lack of information regarding the contribution of run-off flows, the influence of rainfall, unmonitored stretches of the Olifants River and its tributaries as well as periods of poor monitoring of the analytical water quality and flow volumes.

Phosphate pollution mainly occurs due to poor treatment of human sewage and farming activities including the use of fertilizers and poor irrigation practices. As both nitrate and ammonia is measured and reported in mg/ℓ as N for the DWA monitoring points, the combined nitrogen load as N is calculated and used in this report.

The known total N load discharged for the period October 1993 to April 2008 into the Loskop Dam *via* the Olifants River is 2 881 tons. The N load for the same period discharged from the dam (4 992 tons) was calculated as 1 312 tons *via* the Olifants River, 1 721 tons *via* the left and 1 959 tons *via* the right irrigation channels respectively. The actual load of nutrients that enters the Loskop Dam clearly cannot be calculated accurately with the available data.

The available analytical data allows for the nutrient load to be analysed as far as downstream of the Flag Boshielo Dam. However, limited monitoring points are available which allows only the analysis of this data for the Olifants River downstream of the Loskop Dam, upstream of the Flag Boshielo Dam as well as downstream of the Flag Boshielo and downstream of the Renosterkop Dam.

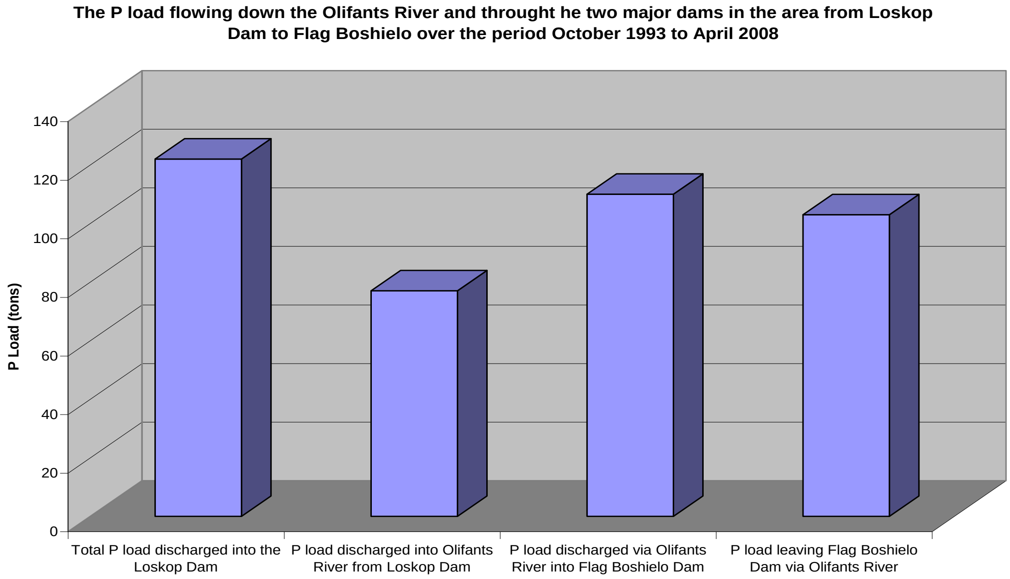

Figure 11: The total P load over the study period flowing down the Olifants River into and out of the Loskop- as well as the Flag Boshielo Dam.

It is clear from this data (Figure 11) that the P load in the Olifants River increases as it flows through the Loskop irrigation scheme when comparing the P load discharged from the Loskop Dam to that entering the Flag Boshielo Dam. The fact that phosphate accumulates in both dams (Figure 11) indicates that the eutrophication potential of both dams are high.

The data for N loads, represented in Figure 12, is very similar as that of the P load for the same area. Clearly a fraction of the N load discharged into the Flag Boshielo Dam remains behind and must be of concern regarding possible eutrophication of this surface water reservoir.

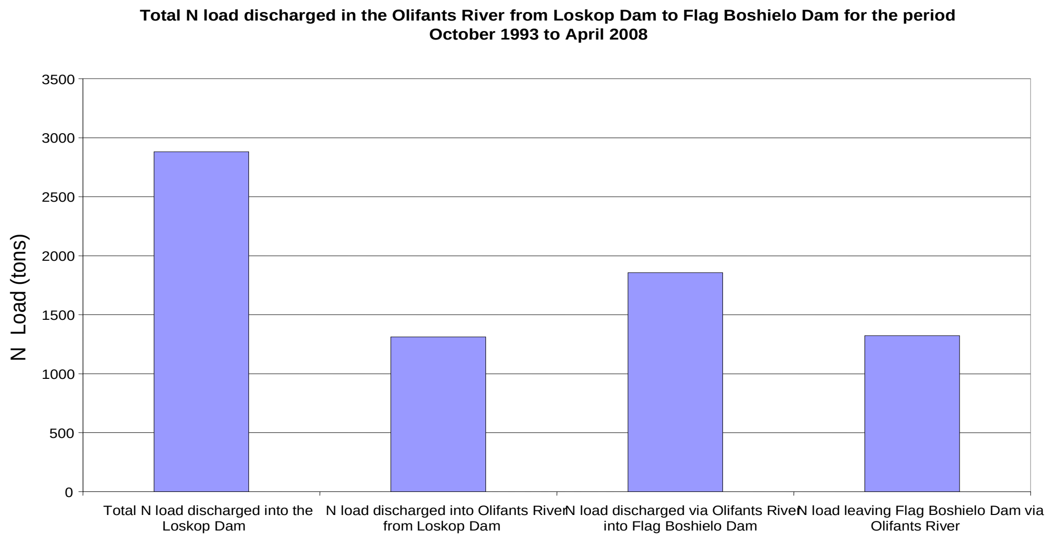

Figure 12: The total N load over the study period flowing down the Olifants River into and out of the Loskop- as well as the Flag Boshielo Dam.

6.3 Direct cost of waste water treatment

The hypertrophic status of the Hartbeespoort Dam is an indication of the severity of the pollution problem associated with industrialization and urbanization in the upper reaches of the Crocodile (West) Marico WMA. This is a clear indication that the pollution prevention measures, including the waste water treatment works (WWTWs) in the study area, are ineffective. Based on the large amount of upgrading of these WWTW's currently underway (see Table 9), to be completed by 2025 at a total cost of approximately R1 364 million, the treatment capacity of these WWTW's appear to be the main concern (Roux and Oelofse, 2010). It remains to be seen whether the ongoing expansion of the waste water treatment infrastructure in this study area will be able to reduce the pollution load flowing into the Hartbeespoort Dam in the long run. If the economic growth and population increase in this area continues at the current rate, the planned increase in waste water treatment capacity will only temporarily alleviate the situation. The question must be asked whether the expansion of WWTW's presents a sustainable solution to the problem of pollution associated with the production of large quantities of waste water in this region.

In study area 1 the capital value for the construction of WWTW's is estimated at R6.5 m/Mℓ/day and the operating cost between R794.1/Mℓ and R1 500/Mℓ (Roux and Oelofse, 2010). It was not possible to do a similar calculation of the costs associated with waste water treatment for study area 2 as the required information requested from the municipalities was not forthcoming

6.4 Direct cost of water purification

Continued discharge of nutrient rich effluent into surface water resources leads to eutrophication, as has occurred in both the Hartbeespoort and Rietvlei dams. Both dams are situated downstream of industrial and urban areas where conventional waste water treatment systems have failed to protect surface water resources from eutrophication. The impact of dam water quality degradation is especially evident at the municipal water purification works facility at Rietvlei Dam. The purification of water to potable standard at this facility has been maintained only through the introduction of continual, increasingly expensive technology upgrades. These technologies are introduced to combat the effects of eutrophication including increased turbidity and the prevalence of blue-green algae (Roux and Oelofse 2010) with its potential to secrete toxic substances into the water. These toxins and their impact on human health, although not included in this study, are summarized well by Messineo *et al.* (2008). Problems regarding the supply of potable water from increasingly polluted water resources are already reported at Hartbeespoort Dam and Brits.

The capital value for the construction of the Rietvlei water purification facility is estimated at R4.6m/Mℓ/day and the operating costs at R1 030/Mℓ for water purification (Roux and Oelofse, 2010). It was not possible to do a similar calculation of the costs associated with water purification for study area 2 as the required information requested from the municipalities was not forthcoming.

According to Quayle *et al.* (2010) study in the Vaal River revealed that the actual chemical treatment costs of producing potable water tend to increase as raw water quality declines (becomes more eutrophic). The cost in 2006 changed from R89.90/Mℓ to R126.31/ Mℓ (Quayle *et al.*, 2010). The same authors estimated the base treatment cost for water to potable standards at the Vaalkop water purification facility abstracting water from Hartbeespoort Dam at R108/Mℓ/day. They further estimated that the introduction of zero-phosphate detergents in the Hartbeespoort dam catchment could result in an annual cost saving of R350 516 at this facility (Quale *et al.*, 2010).

6.5 Cost of eutrophication prevention

The removal of phosphorous at the wastewater treatment works is acknowledged and followed globally, as the most effective eutrophication prevention measure (Frost and Sullivan, 2010). According to Frost and Sullivan (2010), the process to prevent eutrophication in South Africa is threefold:

1. Infrastructure investment to bring WWTW up to normal operations standards
2. Introduction of a biological nutrient removal stage at the WWTW
3. Proper operations of the biological nutrient removal system

The cost of introduction of eutrophication prevention at a typical metropolitan WWTW is estimated at R200 million for the infrastructure upgrade to normal operations standards. An additional R200 million is required for the introduction of biological nutrient removal. The operational cost to implement this system is estimated at R1500/Mℓ (Frost and Sullivan, 2010).

6.6 Cost of eutrophication rehabilitation

The eutrophication levels in the Hartbeespoort Dam are rated as serious and therefore a rehabilitation programme has been put in place to remedy the severity of the problem. The project is valued at R180 million spread over a number of years (Frost and Sullivan, 2010). The proposed interventions and associated costs are outlined in Table 11.

Table 11: Costing of the Hartbeespoort Dam Rehabilitation programme (Frost and Sullivan, 2010)

	Costs (R/annum)
Infrastructure establishment	1 200 000
Algal, Hyacinth, debris and litter removal	1 800 000
Remediation of shoreline and floating islands	2 900 000
Food-web restructuring	2 800 000
Fixed costs sub total	**8 700 000**
Running costs	3 600 000
Management fees	2 400 000
Variable Cost sub Total	**6 000 000**
Grand Total per annum	**14 700 000**

6.7 Discussion and conclusions

The expected economic- and population growth in both study areas predict a further increase in the demand for water and quantities of waste water produced. Therefore, perpetuating the never-ending increase in – and cost associated with the treatment of this increasing volume of waste water. The waste water collection and treatment infrastructure need to keep pace with the increase in waste water generation to prevent further pollution (i.e. to maintain the current poor status quo). The example of Rietvlei water purification works of infrastructure and technological upgrades includes a dissolved air flotation (DAF) system installed in 1980 followed by activated carbon treatment in 1999. More recently ozonation equipment was introduced and as Cyanobacteria, dominating the algal populations in Rietvlei Dam in recent years, complicate treatment and increase associated costs, the "Solarbee" algae management system was recently introduced (Roux and Oelofse 2010). An increase in sophistication of treatment technologies often require operators with higher or different skills levels which could result in higher labour costs at these facilities.

When doing a direct comparison of the cost of waste water treatment and water purification in study area 1 (Table 12), there does not seem to be a major difference in construction and operational cost.

Table 12: Cost comparison of WWTW and WPW

	WWTW (cost/ Mℓ/day)	WPW (Rietvlei) (cost/ Mℓ/day)
Construction	R6.5 million	R4.6 million
Operations	R794.1-R1 500	R1 030

When comparing the cost of rehabilitation of the Hartbeespoort Dam (R180 million) with that of pollution prevention (R400 million per WWTW), it seems cheaper to do the rehabilitation than to prevent the pollution. However, it should be noted that indirect costs have not been taken into account. In addition, the projected eutrophication levels in South Africa suggest that treatment cost may be significantly affected by eutrophication in future (Quayle *et al.* 2010). The question therefore should no longer be around which the cheaper option is, but rather whether or not we will be able to afford the treatment cost into the future.

7 Cost of salinity

7.1 Salinity in study area 2

The Upper Olifants River catchment is characterized by coal mines discharging polluted water into the local streams resulting in local acidification and regional salinisation. Mine water in the Olifants River catchment amounts to only 4.6% of the total water usage but contributes 78.4% of the total sulphate load (Van Zyl *et al.*, 2001). Various levels of treatment are required in order to render the mine water suitable for potential uses at acceptable water sulphate concentrations after treatment. Different uses and acceptable sulphate levels are indicated in Table 13 below.

Table 13: Acceptable sulphate levels for different potential uses in the Upper Olifants Catchment (adapted from Van Zyl *et al.*, 2001).

Water use	Acceptable sulphate level in mg/ℓ
Irrigation	200
Coal processing plant	1000
General industrial use	500
Discharge to public streams	500
Potable use	200
Cooling water in power stations	20-40

The Loskop Valley in the Middle Olifants, downstream of the Loskop Dam has a long history of intensive irrigation farming and a serious salinity problem (Aihoon *et al.,* 1997) as is evident in Figure 13. Although the total dissolved solids (TDS) show an increasing trend over time, the chloride levels remained fairly constant over the same period. Monitoring data of DWA indicate that for the period October 1993 to April 2008, the TDS load in the Olifants River between Loskop Dam and Flag Boshielo Dam increased over distance, downstream. The water from this river is destined for irrigation and to supply potable water to the towns of Groblersdal and Marble Hall and a large number of populated areas along the Olifants and its tributaries (Figure 14). Downstream users impacted include rural domestic users of river water without purification facilities. High levels of TDS may affect the taste of the drinking water and washing may require higher quantities of detergent and soap (Aihoon *et al*., 1997).

The available data also exhibit periods of very high TDS loads at the different monitoring points, probably due to wet and dry seasonal flow patterns for the Olifants River. Periods of high TDS loads for this river system appear to be more severe between 2005 and 2008 than similar high TDS load events that occurred over the period 1993 to 2001. This could indicate that the total TDS load for this river system is increasing and that continuous salinisation of the entire Olifants River system is a very real danger that could affect the quality of both potable water sources available for purification as well as irrigation activities.

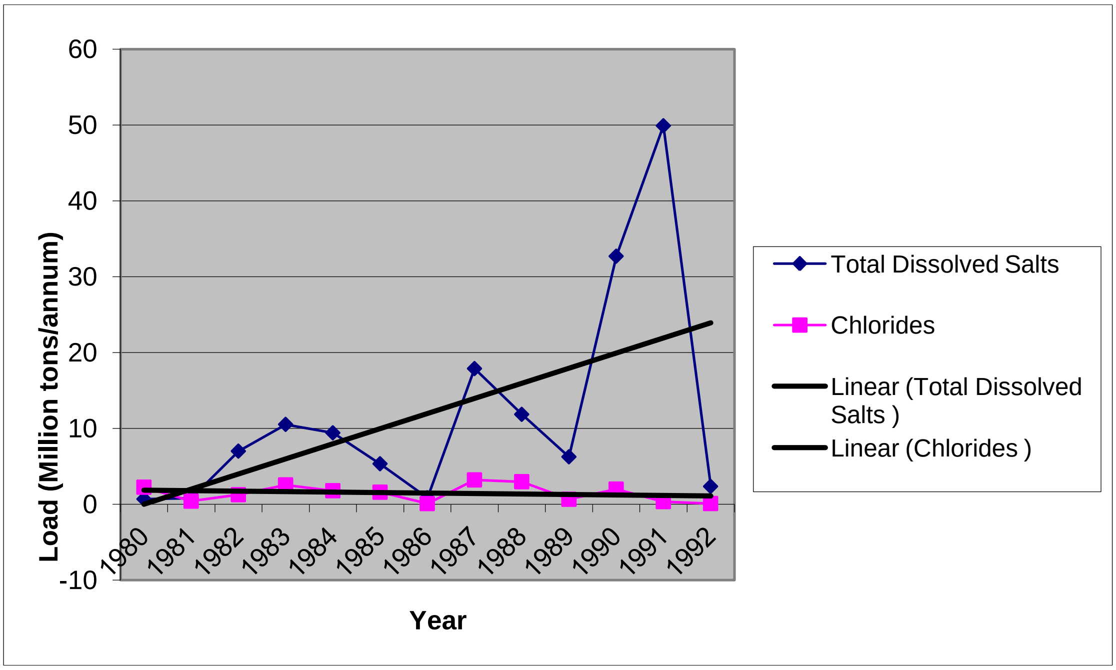

Figure 13: Annual non-point source emissions of salts (TDS) and Chlorides from the Loskop Valley into the Olifants River (data from Aihoon *et al.*, 1997).

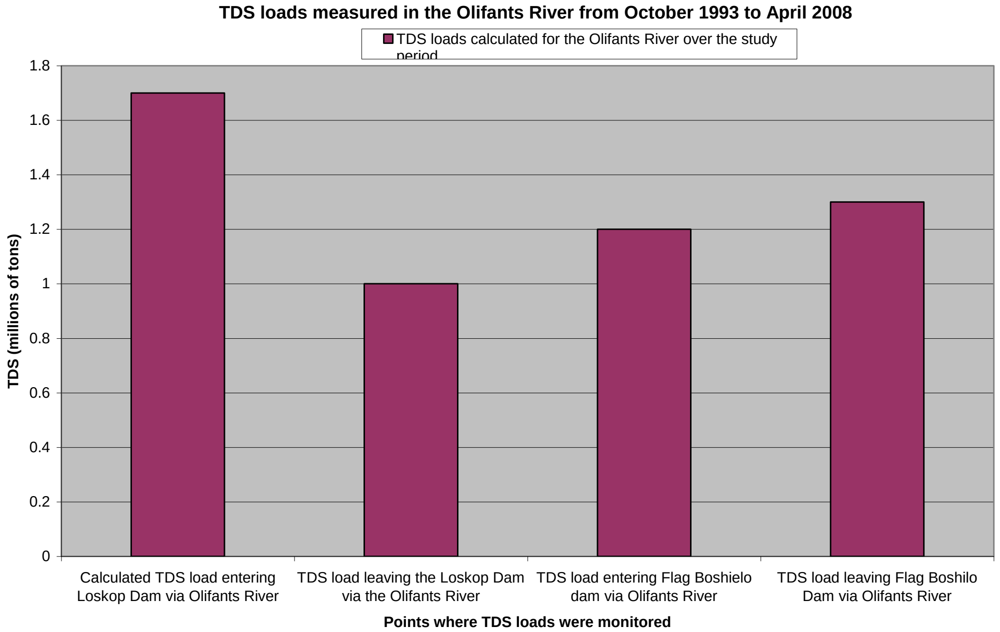

Figure 14: The total TDS loads measured in the Olifants River over the period October 1993 to April 2008

7.2 Cost of salinity to the Loskop Water User Association

This chapter aims to quantify the impact of salinisation on commercial irrigated agriculture within the Loskop Water User Association (WUA) area in monetary terms. A clear distinction should be made between the impacts of water pollution on irrigated agriculture as opposed to the impact that irrigated agriculture may have on water quality. Downstream impacts (externalities) caused by irrigated agriculture will not be covered in this report. The focus here is specifically on the impact of pollution on agriculture, and not vice versa.

The main commercial crops under irrigation in the area include wheat, maize, vegetables, tobacco, peanuts, cotton and citrus. The scheme is operated on a demand scheduling basis and is currently fully allocated, with deficit irrigation fast becoming the norm rather than the exception, because further expansion of the scheme is not possible.

7.2.1 Methodology and data inputs

Amongst the different pollutant categories (microbial pollution, eutrophication, etc), only salinisation has a direct and known impact on agriculture in the Loskop WUA (Aihoon *et al*., 1997, Ferreira, 2009), which makes it possible to quantify the impact of this form of pollution on irrigated agriculture. Salinity was therefore the only pollutant that could be used to estimate the marginal impacts of pollution on irrigated agriculture. This does not imply that the other pollutants have no impact on irrigated agriculture, and this chapter can therefore not be seen as a fully inclusive, comprehensive estimate of the impact of polluted water on agriculture.

The salinity of water in the root zone of commercial crops is determined by various factors, of which the micro-climatic conditions, quality of irrigation water and irrigation management practices are amongst the most important. The quantification of the salinity-yield relationship of a particular crop is critical to determine the monetary impact of salinity on the industry. However, plant responses to known salt concentrations cannot be predicted on an absolute basis, but rather on a relative basis, which provides general salt tolerance guidelines (Maas, 1986). Commercial crops can tolerate salinity increases up to a threshold, from where a linear decrease in crop yield could be expected. Different crops have different thresholds and sensitivity slopes, which means that the tolerance of plants varies among species. Irrigation management practice is the main variable which will determine the probability of reaching these pollution thresholds.

The impact of salinisation on agriculture is quantified in terms of the pollution-yield relationships and the associated impact of water pollution on the profitability of irrigated crops. This relationship has been thoroughly researched for different crops and areas (Urban-Econ, 2000, Viljoen and Armour, 2002, Du Preez *et al*., 2000). The researchers took these relationships as a point of departure for the value estimates in the study area. Only the main irrigated crops which have corresponding pollution-yield relationship data available, were included in the study (shaded crops in Table 14). Again, this study cannot claim a fully inclusive estimate in this regard, because the pollution-yield relationship has not been quantified for all crops in the study area.

The impact of water pollution on commercial irrigation was assessed in terms of the decrease in marginal value product (MVP) of irrigation water for selected crops (Table 15) on typical farm units (25 hectares) in the Loskop WUA. Marginal value product of water for a particular crop can be defined as the additional income being generated for every additional cubic meter of water used to irrigate the particular crop (Nieuwoudt *et al*., 2004). (It should be noted that if the optimum irrigation level is exceeded, the MVP will start declining due to over-irrigation). The reason for opting for MVP is to emphasize the relative change in typical farm income due to increments in the pollution levels of irrigation water. Time series data on pollution levels (TDS) for the Loskop WUA was not available (Ferreira, 2009) and consequently the associated loss in typical farm income per each 100 mg/ℓ increment in TDS was modeled from the salinity threshold point for each selected crop. This implied a modeling increment which ranged from 900-1800 mg/l TDS. Another reason for opting for marginal values is that it explicitly shows value losses due to increments in the pollution levels, whereas average values imply a constant change between different pollution levels (which is not the case in reality). In most policy-related applications of economic valuations involving water, the relevant quantity that needs to be known is the marginal value rather than the average (or total) value of water. The reason is that, given that water is a necessity of life, most people have some access to some water all the time; policy interventions therefore change the quantity and/or quality of access to water, rather than transforming a situation from no access to some access. The point is that there is likely to be some degree of diminishing marginal utility (for users) and diminishing marginal returns (for producers), which then implies that there can be a substantial difference between the marginal value of an increase in water supply and its average value. This needs to be emphasized because researchers tend to prefer to use average values to measure the benefits of a policy intervention, while marginal values should be preferred (Hassan and Farolfi, 2005, Lange and Hassan, 2006, Young, 2004). Irrigation management practice is the main variable which will determine the probability of reaching these pollution thresholds.

Table 14: Theoretical threshold (TDS mg/ℓ) for commercial crops in South Africa (Du Preez *et al*., 2000)

Crop	Salinity threshold (TDS mg/ℓ)	Crop	Salinity threshold (TDS mg/ℓ)
Strawberry	650	Cabbage	1170
Green beans	650	Celery	1170
Carrots	650	Lucerne	1300
Aubergine	715	Spinach	1300
Onions	780	Cucumber	1625
Radish	780	Broccoli	1820
Citrus	845	Rice	1950
Lettuce	845	Peanuts	2080
Plums	975	Peas	2210
Almonds	975	Fescue	2535
Grapes	975	Beetroot	2600
Sweet potato	975	Asparagus	2665
Pepper	975	Gemsquash	3055
Clover	975	Soya	3250
Apricots	1040	Ryegrass	3640
Peach	1105	Wheat	3900
Potatoes	1105	Triticale	3965
Garlic	1105	Sorghum	4420
Maize	1105	Sugarbeet	4550
Sweet corn	1105	Cotton	5005
Sugarcane	1105	Barley	5200
Tomatoes	1105	Rye	7410

Representative crops for the study area were selected in consultation with the Loskop WUA manager (Ferreira, 2009); also see Appendix 1. These crops have been used to present the impact of salinity on MVP for a typical farm size (being 25 hectares) in the Loskop WUA. Coverage data was verified with Statistics South Africa data (2002). The next step was to derive salinity thresholds and sensitivities (i.e. rate of yield decline with increments in salinity) for the selected crops. This was done based on available literature (Aihoon *et al*., 1997, Maas *et al*., 1983, Maas, 1986, Armour, 2007, Armour and Viljoen, 2000, Viljoen and Armour, 2002) and the analysis was scaled between 900 mg/l to 1800 mg/l TDS (the reasons for this range will be explained in the assessment of results). Crop enterprise budgets were obtained for geographically comparable areas (Western Cape Department of Agriculture, 2006, Van Zyl, 2009, Deciduous Fruit Producers Trust, 2008) to be used as inputs in the MVP estimates. The following input variables were extracted: yield per hectare; crop producer prices; typical enterprise size in the Loskop WUA; total variable production cost per hectare; fixed production cost per hectare. Data was inflated using the producer price index of 1980 until 2008 and researchers used a 7.78% discount rate for maize, wheat and citrus (current prices were used for potatoes). SAPWAT (Crosby, 1996, Van Heerden *et al*., 2008) simulations were employed to estimate the water requirement per hectare per year (Appendix 3). A water constraint of 7700 cubic meter per hectare per year was used (Ferreira, 2009) (see Appendix 1). This data was used to develop a matrix for the linear programming (LP) model with an objective function and constraint functions which were transferred onto data sheets and eventually onto cards for the LP solution. Simulations were run to obtain the optimal enterprise solutions for the selected crops at increasing salinity levels.

The pollution-yield relationships used for estimating the MVP of water for different pollution levels are summarized in Table 15.

Table 15: Yield response relationships used for estimating the marginal product value of water for different pollution levels. Data adapted from (Urban-Econ, 2000, Viljoen and Armour, 2002, Du Preez *et al*., 2000, Maas, 1986).

Crop	Typical yield (t/ha)	Threshold salinity (TDS in mg/ℓ)	Yield responses (% of unconstrained yield) for different salinity levels (TDS in mg/ℓ)									
			900	1000	1100	1200	1300	1400	1500	1600	1700	1800
Maize	7.87	1207	100	100	100	100	98.5	97	95	93.5	92	90
Wheat	5.47	4346	100	100	100	100	100	100	100	100	100	100
Potatoes	37.8	1151	100	100	100	99	97	94.5	92	90	88	86
Citrus	45	975	100	96	82	67	53	38	24	9	0	0

The above-mentioned data was combined with standardised crop budgets for each of the selected crops (see Appendix 2) to present the monetary impact of salinity for these crops. A total of 25 MVP simulation runs were completed to cover all the different crop and water pollution combinations for the study area.

7.2.2 Assessment of modelling results

Modelling results are illustrative, give a relative perspective and are dependent on certain assumptions. The results as presented here should therefore not be used to make decisions concerning farming operations e.g. uproot citrus trees when TDS reach 1200 mg/ℓ. For the purpose of the MVP estimates, it was assumed that the salinity of irrigation water is directly proportional to the salinity of the saturated soil (which is not always the case). The average maximum allowable salinity in the Loskop area is 1 700 mg/ℓ TDS, while the recommended operational salinity limit is a maximum of 1 000 mg/ℓ TDS (Ferreira, 2009). These specifications determined the salinity range that was used to investigate economic impacts of salinisation in this study. The range from 900 mg/ℓ TDS (100 mg/ℓ TDS below the recommended salinity limit) up to 1 800 mg/ℓ TDS (100 mg/ℓ TDS above the maximum allowable water salinity) was used to present a realistic picture of the situation in the Loskop WUA area.

Table 16: Irrigation water productivity at different water pollution levels

	MVP in R/m^3			
TDS (mg/ℓ)	Maize	Wheat	Potatoes	Citrus
900		0.45		3.65
1000		0.45		3.37
1100		0.45	5.60	2.38
1200	0.47	0.45	5.46	1.32
1300	0.44	0.45	5.17	0.33
1400	0.41	0.45	4.82	0.00
1500	0.37	0.45	4.47	
1600	0.34	0.45	4.18	
1700	0.31	0.45	3.90	
1800	0.27	0.45	3.62	

The relationship between the saturated soil salinity and the MVP of saline irrigation water as applied to the four selected crops under investigation is presented in Table 16. All the investigated crops' MVP's were observed to decline with increasing salinisation within the salinity range of the study (900-1 800 mg/ℓ TDS), except that of wheat which remained at R0.45/m^3 throughout the salinity range. Wheat, being salinity-tolerant, had the highest salinity threshold of 4 346 mg/ℓ TDS, and the gradient of MVP of its irrigation water as salinity increased was thus zero. MVP of maize was observed to decline from R0.47/m^3 to R0.27/m^3 of water as salinity increased from 1 200 to 1 800 mg/ℓ TDS. The gradient of MVP of its irrigation water as salinity increased was -0.033. The MVP of potatoes was observed to decline from R5.60/m^3 to R3.62/m^3 with a salinity increase from 1 100-1 800 mg/ℓ TDS. The gradient of MVP of its irrigation water as salinity increased was -0.29. The MVP of oranges was observed to decline from R3.65/m^3 to zero with a salinity increase from 900-1 400 mg/ℓ TDS. The gradient of MVP of its irrigation water as salinity increased was -0.81. Potatoes had the highest MVP per m^3 of irrigation water used, followed by citrus, maize and wheat. The MVP's of the four crops were respectively R5.56/m^3, R3.65/m^3, R0.47/m^3 and R0.45/m^3 of irrigation water before the salinity threshold started to have an impact on the values.

Table 17: Optimal farm productivity at different water pollution levels

TDS (mg/ℓ)	Gross margin above specified cost for a monoculture 25 ha farm unit (R)			
	Maize	Wheat	Potatoes	Citrus
900		83179		678248
1000		83179		623850
1100		83179	775994	433456
1200	69015	83179	755984	229463
1300	64499	83179	715964	39069
1400	59983	83179	665940	25247
1500	53961	83179	615915	
1600	49445	83179	575896	
1700	44929	83179	535877	
1800	38908	83179	495857	

The objective of the linear programming model is to maximize net returns from agricultural production, subject to production and resource constraints on water and land. The model optimises for a monoculture and assumes a perfectly competitive market structure, and linearity in production relationships (Jabeen *et al*., 2006). The optimal solutions for the four selected crops along the salinity range of 900-1 800 mg/ℓ TDS is presented in Table 17. The total gross margin above specified costs (TGMASC) values represents the optimal solution at each increasing TDS level. Potato had the highest TGMASC before its salinity threshold level of R775 994, followed by orange with R678 248, then wheat with R83 179 and finally maize with R69 015.

7.2.3 Discussion and limitations

This chapter presented the monetary impact of water pollution in terms of the income being lost when crops are irrigated with polluted water, viz. the production costs associated with clean versus salinised irrigation water. A key step to achieve this was to assess how the quality of irrigation water changed "with" and "without" salinisation prevention measures. The results of the assessments provided estimates for the baseline (without policy) water quality, and the subsequent (with policy) water quality. Given estimates of the baseline and the subsequent groundwater conditions, the change in the irrigation water quality was defined.

The MVP of irrigation water represents the "true" economic value of an additional unit of irrigation water to a farmer. It represents the "economic value in use" to the farmer. Generally speaking, this additional unit of water would in turn produce additional agricultural output, whose value is dependent upon the type of crop grown and the producer price that is specific to the region (Jabeen *et al*., 2006). The MVP gradient of irrigation water represents the economical sensitivity of a crop to salinisation (Figure 15). Citrus, being the most sensitive, has an MVP gradient of -0.81, which implies that for every 100 mg/ℓ TDS increase in salinity, citrus production suffers a R0.81/m^3 loss in the MVP of irrigation water. Potato follows with an MVP gradient of -0.29, this implies that for every 100 mg/ℓ TDS increase in salinity, potato production suffers a R0.29/m^3 loss in the MVP of irrigation water. Maize had a MVP gradient of -0.033, which implies that for every 100 mg/ℓ TDS increase in salinity, maize production suffers a R0.033/m^3 loss in the MVP of irrigation water. Wheat, being the most salt tolerant, did not display any loss in the MVP of irrigation water within the TDS investigation range of this study (900-1 800 mg/ℓ TDS).

Apart from being the most sensitive to salt in economic terms, citrus production was also observed to suffer the highest decrease in terms of the MVP per m^3 of irrigation water as salinity increased. The MVP of citrus decreased from R3.65/m^3 at 900 mg/ℓ TDS to R0.00/m^3 at 1 400 mg/ℓ TDS, amounting to a R3.65/m^3 MVP loss for irrigation water. R3.65/m^3 represents the costs associated with a change in irrigation water quality "with" and "without" salinisation prevention strategies and policies in play. It is a comparison of the MVP of clean irrigation water versus the MVP of salinised irrigation water within the salinity investigation range of this study. The difference is the loss in the economic value of irrigation water in a situation where salinisation exists, in contrast with a situation of clean irrigation water.

The linear model applied an irrigation water availability constraint of 7 700 m^3/ha for all crops. Citrus required 10 510 m^3/ha of irrigation water, which was in excess of the water constraint, which dictated that only 18.32 ha of the typical farm unit could be irrigated. The high sensitivity of citrus to salt resulted in the MVP/m^3 of irrigation water reaching zero at a salinity level of 1 400 mg/ℓ TDS. At this level, it was economically not viable to produce any citrus. A typical citrus producing farm in the Loskop WUA area was observed to reap a TGMASC ranging from R678 248 to R-25 247 as salinity increased; this translated to a loss in TGMASC of R703 495 per typical farm in the Loskop WUA area as a result of salinisation. A typical maize producing farm in the Loskop WUA area was observed to reap a TGMASC ranging from R69 015 to R38 908 as salinity increased; this translated to a loss in TGMASC of R30 107 per typical farm in the Loskop WUA area as a result of salinisation. The MVP of maize decreased from R0.47/m^3 to R0.27/m^3, which is a R0.20/m^3 MVP loss for irrigation water. R0.20/m^3 represents the costs associated with a change in irrigation water quality "with" and "without" a salinisation prevention policy in place. A typical potato producing farm in the Loskop WUA area was observed to reap a TGMASC ranging from R775 994 to R495 857 as salinity increased, this translated to a loss in TGMASC of R200 137 per typical farm in the Loskop WUA area as a result of salinisation. The MVP of potato decreased from R5.60/m^3 to R3.62/m^3, translating to a R1.98/m^3 MVP loss for irrigation water. R1.98/m^3 represents the costs associated with a change in irrigation water quality "with" and "without" a salinisation prevention policy in place. A typical wheat producing farm in the Loskop WUA area was observed to reap a constant TGMASC of R83 179 as salinity increased. Wheat is tolerant to salinity, and as such did not show any economic losses due to salinisation pollution within the salinity range of this study. The MVP of wheat remained constant at R0.45/m^3, there was therefore a R0.00/m^3 MVP loss for irrigation water.

Only for wheat there was no cost associated with a change in irrigation water quality "with" and "without" a salinisation prevention policy in place in the Loskop WUA area.

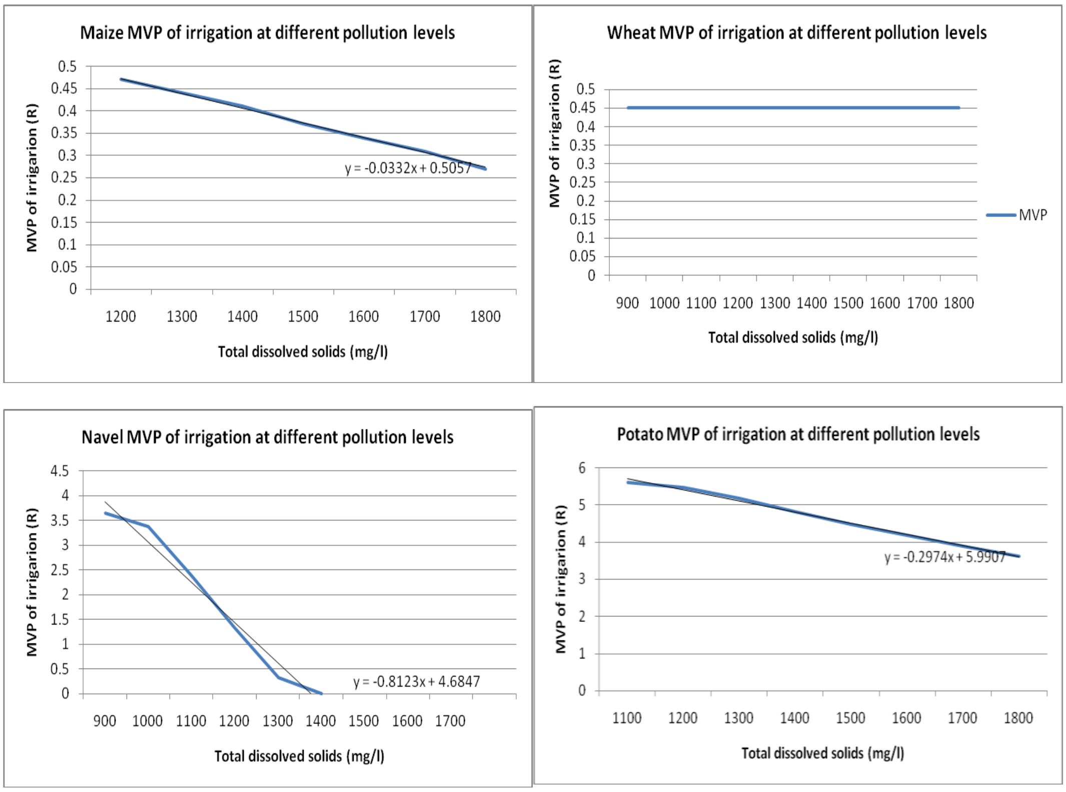

Figure 15: MVP of selected crops at different pollution levels

Upon comparing the MVP figures with the current water tariff facing farmers in the Loskop WUA area of R0.14/m^3, the results of this study revealed that water is generally underpriced. Water is regarded as a production factor (intermediate good), and the benefits it generates to the farmer (value in use) ought to be reflected in the price (the value in exchange). The MVP's calculated in this study were all above the general tariff level, and in order to achieve the societal goal of allocation efficiency, the MVP (economic value in use) should be more or less equated to the price paid (value in exchange). Therefore, in theory, ample room exists to increase the price of water for these four crops. However, it may not be feasible or fair to expect of farmers to carry the full burden of such price increases. One limitation of the method as used in this chapter is that it assumes a constant conversion factor across different soil types, drainage statuses and irrigation systems used. This is not the case in practice. Consequently, on-farm modeling should be done with leaching fraction models (Armour, 2007), which are calibrated for the specific farm.

7.3 Cost impact of treating salinity from mining operations

Treatment of mine effluent is seen as a pollution prevention measure. According to Van Zyl *et al.* (2001) the capital cost of desalination treatment alternatives varied between R4 million/Mℓ/day and R10 million/Mℓ/day and the running cost between R2/m^3 and R5/m^3. They estimated that limestone and lime treatment is the most cost effective technology for neutralization and partial sulphate removal of acidic or sulphate-rich water to sulphate levels of less than 1500 mg/ℓ. The estimated capital and running cost for the collection and treatment of excess mine water to a quality suitable for selected urban and industrial applications amounted to R528.5 million and R55.7 million/year respectively. Collection and treatment of mine water to a quality suitable for irrigation amounted to R68.2 million and R11.9 million/year (Van Zyl *et al*., 2001).

7.4 Cost of salinity in the Vaal River system

A comparison of the direct cost impacts of salinity on the entire economy in the Vaal River system was done in 2000 (Urban-Econ, 2000). The average salinity in the Vaal River at the time was 500 mg/ℓ. Cost of salinity below 500 mg/ℓ implies a saving while salinity above this level will result in an increase in the direct cost as a result of increased salinity. The results of the 2000 study are provided in Table 18.

Table 18: Direct cost of salinity in the Vaal River (1995 Values in million Rand) (Urban-Econ, 2000)

Sector	**Salinity mg/ℓ TDS**					
	200	**400**	**600**	**800**	**1000**	**1200**
Mining	(7.309)	(2.212)	0.844	4.863	10.209	17.816
Business and services	(1.843)	0.487	1.211	1.707	2.209	2.697
Manufacturing 1	(0.145)	0.028	0.086	0.123	0.160	0.198
Manufacturing 2	(2.825)	0.294	1.351	1.993	2.635	3.278
Agriculture	0.000	0.000	0.439	0.439	0.427	0.503
Households (suburban)	(35.121)	(11.707)	11.707	35.121	58.535	81.949
Household (township)	(27.927)	(9.309)	9.309	27.927	46.544	65.162
Household (informal)	(5.081)	(1.694)	1.694	5.081	8.469	11.855
Totals	(80.251)	(24.113)	26.640	77.253	129.225	183.457

The direct costs are however a poor reflection of the total cost impacts of salinity. At relatively low levels of salinity it is the community and other service sectors that will be most adversely affected. At high levels of salinity the gold mining sector in the Vaal River system will have to incur the highest cost to combat salinity (Urban-Econ, 2000).

7.5 Discussion and conclusions

Farmers in the Olifants River catchment suffer losses in TGMASC as a result of increased salinity. Depending on the crop under irrigation, this value ranges between R30 107 (maize) to R703 495 (citrus) annually per typical farm (25 ha) in the WUA. Thus, considering this typical farm size and given that the irrigation area in question is 16 136 ha, there are in the order of 645 farms which implies an estimated total loss in gross margin above specified costs as a result of increased salinity in the Loskop WUA of between R19 million and R454 million per annum. The cost of collection and treatment of mine water in the Olifants River catchment, to a quality suitable for irrigation purposes, will cost between R11.9 million and R68.2 million/annum (Van Zyl *et al*., 2001) (Table 19).

Table 19: Cost comparison of increased salinity to farmers and prevention of salinity by mine water treatment to irrigation standards

	Low value	High value
Cost of salinity in Loskop WUA expressed as losses in TGMASC	R19 million	R454 million/annum
Cost of prevention at mines (Treatment to irrigation standards)	R11.9 million	R68.2 million/annum

The cost implications to mines to prevent salinisation of water resources by the mine effluent is therefore likely to be less than the potential losses farmers in the Loskop WUA could suffer as a result of increased TDS levels in irrigation water. The cost calculations do not include the loss of food crops and therefore food security and the associated social cost if salinisation continues to increase.

8 Cost of microbial pollution

Healthcare expenditure by a household experiencing illness in rural South Africa, incurred a direct cost burden of 4.5% of total household expenditure in 2009 (Goudge *et al.*, 2009). In addition, a visit to a public clinic generated a mean cost burden of 1.3%; 20% of households incurred a burden of over 10% for complex treatments and transport costs accounted for 42% of this burden (Goudge *et al.*, 2009). An outpatient visit generated a burden of 8.2%, while an inpatient stay incurred a burden of 45%. About 38% of individuals who reported illness did not take any treatment action (Goudge *et al.*, 2009). This is not surprising when considering the high levels of unemployment and poverty in rural settings.

In the past decade, much evidence has emerged that supports the beneficial outcomes of securing safe potable water, sanitation, and hygiene in developing countries. Many studies link improvements in sanitation and provision of potable water with dramatic reductions in waterborne morbidity and mortality. For example, a review in 1991 of over 100 studies of the effects of clean water and sanitation on human health found that the median reduction in deaths from waterborne diseases was 69% among people with access to potable water and proper sanitation (Hinrichsen *et al.*, 1997). In the U.S. and Central Europe, water and hygiene-related diseases have been significantly reduced since the start of the 20th century after the installation of water supply and sanitation (WSS) systems. A meta-analysis of the impact of such interventions on gastroenteritis concluded that increasing water quantity reduced the occurrence of diarrhoea by 25%, whereas point-of-use household water treatment (water methods used to improve the quality of water at the point of consumption) reduced the occurrence by 35%, while improved sanitation led to reductions of approximately 32% in occurrence (Montgomery and Elimelech, 2007). Furthermore, sanitation and point-of-use interventions may have resulted in greater reductions because they directly block pathways of exposure. The meta-analysis, and evidence based on the trajectory that developed countries have gone through, provides evidence that water pollution is a major contributory factor to waterborne illness. However, in developing countries, water and sanitation systems are still severely lacking. As a result, millions suffer from preventable illnesses, while deaths are common (Montgomery and Elimelech, 2007). Thus, there is a significant burden of disease (BoD) in the form of mortality, morbidity and the associated economic burden that could be prevented (also in South Africa) through tighter control of microbial pollution and its contributing environmental factors (CSIR, 2010).

8.1 Microbial water quality in study area 2

The National Microbiological Monitoring Programme (DWAF, 2002c) developed a simple screening method to identify the risk of faecal contamination in various catchments in South Africa. This screening method confirmed that the highest faecal contamination rate is derived from high population densities with poor sanitation services. Based on this information a potential surface faecal contamination map for the whole of South Africa using national databases for population density and degree of sanitation was developed (DWAF, 2003c), and applied to assess the risk of microbial contamination of surface water and groundwater resources in South Africa. The portion applicable to the surface water in Olifants WMA, given in Figure 16, shows the low, medium and high risk areas in the WMA.

The upper Olifants River catchment shows a low potential for microbial contamination. The Elands River catchment and the Middle Olifants River have a high potential for contamination due to insufficient water and sanitation services in the densely populated areas of the former Lebowa homeland. Phalaborwa area also shows a high potential due to the mining activities in the area (DWAF 2003c).

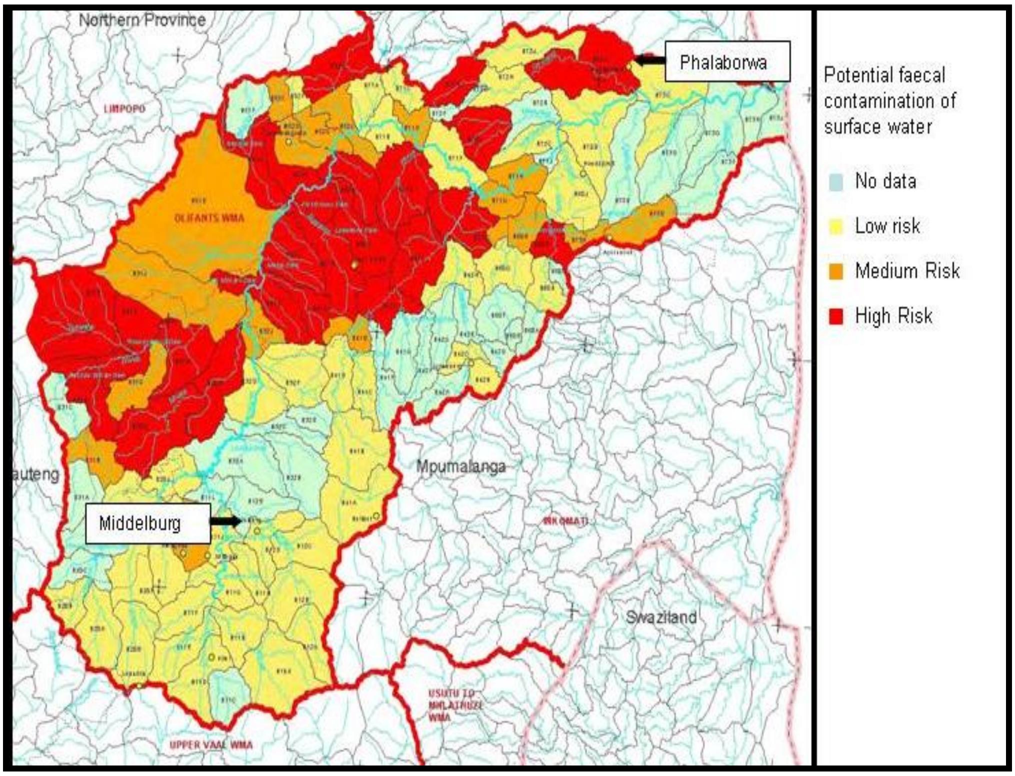

Figure 16: The potential faecal contamination of surface water in the Olifants WMA ranked through low, medium and high risk. (Adapted from DWAF 2003c)

8.2 Costs of microbial pollution in study area 2

The direct costs of diarrhoea, based on estimated diarrhoea incidences and actual diarrhoea deaths in the study area, will be determined.

8.2.1 Cost of illness (COI) approach

In recent years, there has been an unabated rise in health care costs globally. It has therefore become useful to calculate these costs and describe those (Heijink *et al.*, 2006). Cost of illness (COI) studies can be used for this purpose. COI studies illustrate the economic burden of diseases in a country but can also be applied to do cross-country comparisons (Heijink *et al*., 2006).

The economic burden associated with diarrhoea includes all direct and indirect costs associated with the disease. Direct medical costs include medication, diagnostics, personnel and hospital bed-day costs endured by providers, patients and caregivers while non-medical direct costs include transport costs borne by patients and or caregivers. Indirect costs on the other hand include time lost from productive work borne by patients/caregivers and/or society as well as the additional health care costs during a prolonged lifetime after treatment and recovery from the illness (Heijink *et al*., 2006; WHO, 2005).

Population figures are readily available from Statistics South Africa for local municipal areas, districts and provinces within South Africa. This is not the case for water management areas in the country. The most recent mortality (death) statistics available for South Africa is for 2007. This study therefore made use of the 2007 population figures to determine the number of people in the WMA.

The study assumed an equal distribution of people over each local municipal area and made use of the percentage area (km^2) within the WMA (from the GAP (2004) analysis data clipped by means of GIS capabilities[1]) to determine the size of the population within each local municipal area within the Olifants River WMA. Based on these estimates, in 2007, the Olifants River catchment supported approximately 3.4 million people.

Table 20 summarises the population at local, district and provincial level within the catchment area. The biggest proportion of the population within the catchment forms part of the Limpopo Province (1 799 713), followed by Mpumalanga Province (1 443 547). The Greater Sekhukhune District within Limpopo Province is the only district which falls 100% within the WMA. The Nkangala District within the Mpumalanga province supports the largest population within the WMA (1 202 175).

Table 20: Population within the Olifants Water Management Area, 2007

Province	2007 Population WMA
GAUTENG	**144 636**
DC42: Sedibeng	1 850
Lesedi	1 850
DC46: Metsweding	108 263
Kungwini	84 087
Nokeng tsa Taemane	24 176
EKU: Ekurhuleni	34 522
Ekurhuleni	34 522
LIMPOPO	**1 799 713**
DC33: Mopani	256 330
Ba-Phalaborwa	76 904

[1] The study attempted using the actual population density figures from the 2004 GAP analysis for each mesozone, but there were discrepancies between the GAP population figures and that of the StatsSA 2007 data for which could not be explained.

Province	2007 Population WMA
Greater Tzaneen	83 570
Maruleng	95 779
(blank)	77
DC35: Capricorn	418 276
Lepele-Nkumpi	234 225
Polokwane	184 051
DC36: Waterberg	34 683
Bela-Bela	18 929
Modimolle	401
Mogalakwena	2 631
Mookgopong	12 723
DC47: Greater Sekhukhune	1 090 424
Elias Motsoaledi	247 488
Fetakgomo	112 232
Greater Marble Hall	124 510
Greater Tubatse	343 468
Makhuduthamaga	262 726
MPUMALANGA	**1 443 547**
DC30: Gert Sibande	118 995
Albert Luthuli	568
Govan Mbeki	104 890
Msukaligwa	13 537
DC31: Nkangala	1 202 175
Delmas	47 967
Dr JS Moroka	244 155
Emalahleni	435 217
Highlands	14 042
Steve Tshwete	182 277
Thembisile	278 517
DC32: Ehlanzeni	122 377
Bushbuckridge	69 661
Thaba Chweu	51 887
(blank)	828
Total 2007 Population in WMA	**3 387 896**

The young and elderly, as well as the poor are usually the most affected by diarrhoea. Although diarrhoea is not a notifiable disease in South Africa incidence of children under five are recorded when they report for treatment at local municipal health clinics or hospitals. Diarrhoea incidence of adults treated for diarrhoea at hospital or if they died of the disease is recorded. Diarrhoea incidence data for South Africa and for the WMA are therefore expected to be grossly underestimated.

HIV/AIDS has a huge impact on the diarrhoea incidence as well as mortality, also in the adult population of the country and within the study population as seen from the increase of diarrhoeal deaths in older age groups over the last ten years (1997-2006). It is therefore expected that the economic costs due to diarrhoea might be an underestimation of the direct health costs associated with the disease in the area.

Similar to the first ever South African COI study done by Pegram *et al.* (1998) and the follow up study by the Department of Water Affairs (DWAF, 2001), the population within the WMA was grouped into different age groups. In addition to the age groups used by Pegram *et al.* (1998), this study included an additional age group**.**

Table 21 shows the age groups and the reason for categorising the population into the different age groups.

Table 21: Description of age groups used for study

Age group	Description
< 5	Represents infants and children under five, mostly affected by diarrhoea
5-14	Represents a school going child
15-64	Represents the economically active population
> 65	Represents pensioners / retired population and a vulnerable subpopulation in terms of diarrhea

Children under five and people older than 65 are very susceptible to diarrhoea while the other two age groupings were chosen to highlight the possible productivity losses for school going children as well as the adult population in the catchment.

Safe water, improved sanitation and basic household hygiene play an integral role in reducing the diarrhoeal disease burden of South Africa and the WMA specifically.

Many areas within the Olifants WMA are former homeland areas which historically have not been served with basic services. Major improvements have been made by the Department of Water Affairs since 1994 and more recently to supply most areas of the country with safe water, also within this catchment. However, there are still people who do not have access to safe water and improved sanitation services resulting in the use of untreated water sources for domestic purposes. Many of the communities, especially in the rural and peri-urban areas are supplied with treated water supplies via communal or yard taps. Water is then stored in containers for use in the homes. Research has documented the microbiological deterioration of water quality when stored. Pegram *et al.* (1998) also refers to the importance of quantity of water in the prevention of diarrhoea.

For this study, access to water supply and sanitation was (very similar to the approach of Pegram *et al.* (1998)) categorised into two major types:

> RDP Water and Sanitation – which represents a reliable supply of water directly to the household (inside the house or yard taps) with sanitation ranging from at least a Ventilated Improved Pit latrine (VIP) per household to on-site flushing sanitation, such as waterborne sewerage or septic tanks.

< RDP Water and Sanitation – which for this study represents any water sources ranging from untreated water to treated tap water supplied via communal taps (within or further than 200 m) while sanitation ranged from no form of sanitation to at best pit latrines.

This was done to reflect the potential difference in diarrhoea incidence rates and therefore the different cost scenarios for people with varying levels of access to water supply and sanitation. Other confounding factors that could potentially contribute to this and which were not included in the cost calculation include aspects such as type of housing, education, nutritional status as well as household income levels. Another aspect that plays a major role in diarrhoea costs are the access to health services.

Table 22 shows the varying access to water supply and sanitation within each of the three provinces as well as for the entire WMA. The level of access has also been categorised according to the different age groups.

Table 22: Population per age group and access to water and sanitation

Province	**2007 Population spread per age group**			
GAUTENG	**< 5**	**5-14**	**>15-<65**	**>65**
% age group spread	**9.72**	**15.7**	**70.04**	**4.54**
< RDP Water & Sanitation	4 203	6 789	30 287	1 963
> RDP Water & Sanitation	9 855	15 919	71 016	4 603
LIMPOPO	**< 5**	**5-14**	**>15-<65**	**>65**
% age group spread	**11.56**	**25.39**	**56.66**	**6.38**
< RDP Water & Sanitation	169 344	371 942	830 021	93 462
> RDP Water & Sanitation	38 703	85 005	189 696	21 360
MPUMALANGA	**< 5**	**5-14**	**>15-<65**	**>65**
% age group spread	**10.95**	**22.52**	**61.95**	**4.57**
< RDP Water & Sanitation	78 338	161 112	443 201	32 695
> RDP Water & Sanitation	79 730	163 975	451 077	33 276
WMA	**< 5**	**5-14**	**>15-<65**	**>65**
% age group spread	**10.74**	**21.20**	**62.88**	**5.16**
< RDP Water & Sanitation	238 886	471 472	1 398 258	114 811
> RDP Water & Sanitation	125 087	246 875	732 164	60 118

8.2.2 Diarrhoea incidence in the Olifants WMA

The COI approach based on previous studies (Pegram *et al*., 1998; DWAF, 2001) calculated incidence rates for varying levels of water supply and sanitation access for the population within the WMA (Table 23).

Based on recorded diarrhoeal death statistics (StatsSA, 2008), the number of mild, moderate and severe cases were determined according to the Pegram *et al.* (1998) paper.

It was assumed that each person within the catchment would have at least one diarrhoea episode per year. The highest incidence rate used is 2.5 episodes per child for children under the age of five years for the population with less than RDP water supply and sanitation services. This is a very conservative figure since Wright *et al*. (2006) have found that children in rural areas of Limpopo province had up to 7.2 episodes per year. However, this figure was calculated by Pegram *et al*. (1998) based on actual diarrhoea incidence among different age groups in a study area in KwaZulu-Natal with a similar demographic and socio-economic background as the Olifants Water Management Area.

Table 23: Diarrhoeal incidence per age group for the Olifants WMA

Age group	< RDP Water and sanitation	Diarrhoea cases per 1000	Actual diarrhoea incidence
< 5	238 886	2 515	600 798
5-14	471 472	1 001	471 943
>15-<65	1 398 258	750	1 048 693
>65	114 811	1 001	114 925
Total	**2 223 426**	**1 001**	**2 225 650**
Age group	**> RDP Water and sanitation**	**Diarrhoea cases per 1000**	**Actual diarrhoea incidence**
< 5	125 087	501	62 669
5-14	246 875	250	61 719
>15-<65	732 164	104	76 145
>65	60 118	181	10 881
Total	**1 164 244**	**181**	**210 728**

The total deaths per province within the WMA for each age group is summarised in Table 24. The recorded death data for the provinces were apportioned to the WMA and used to calculate the severe cases and therefore the associated treatment and transport costs.

Table 24: Diarrhoeal deaths recorded per province within the Olifants WMA

Age group	Gauteng	Mpumalanga	Limpopo	Olifants WMA
< 5	76	1 549	2 098	3 723
5-14	19	387	525	931
>15-<65	47	1 074	1 396	2 516
>65	8	198	292	498
Total	**150**	**3 208**	**4 310**	**7 668**

8.2.3 Direct costs of diarrhoea in the Olifants WMA

This section follows the COI approaches of Pegram *et al.* (1998) and that of DWAF (2001) to calculate the direct cost of diarrhoea in the Olifants WMA. The approach used for this study was very conservative and only calculated the direct health treatment costs due to diarrhoea as well as the direct transport costs incurred to seek treatment.

Only a small percentage of the total diarrhoea cases require formal treatment or health intervention. The majority of cases is mild and can usually be treated at home. The DWAF (2001) study estimated that approximately 8% of all diarrhoea cases in the areas with below RDP water and sanitation services require treatment, while only 5% of the cases in the areas with above RDP water and sanitation services need treatment.

This study used the direct treatment costs calculated for the DWAF (2001) study which was based on the severity of the disease, e.g., mild, moderate, and severe. Transportation costs to and from medical treatment centers, e.g., health clinics or hospitals, were also considered. It was assumed that these costs were incurred for half (50%) of the trips to the local health practitioner or clinic, while 70% of hospital cases were assumed to make use of public or private transport, with the remainder walking to health services (Pegram *et al*., 1998).

Actual healthcare costs were not available for the study area. The DWAF (2001) study however estimated the cost of treatment for diarrhoea from areas with below RDP water and sanitation services at R1 904 per treatment. Similarly, the average medical treatment costs in areas with above RDP water and sanitation services was estimated to be R1 692 per treatment.

This study assumed that the above-mentioned would also be a true reflection for the Olifants WMA. The consumer price index (CPI) was used to inflate these estimates to current prices in the WMA. The CPI since 1996 is reflected in Table 25. It shows that the average cost of treatment in high service areas for 2007 were R3 349 while average treatment costs in low service areas were R3 769.

Table 25: Consumer Price Index and changes over the years since 1996 (base year = 2008)

Year	Average Inflation rate / year (%)	< RDP service areas (R)	> RDP service areas (R)
1996	7.3	2 043	1 816
1997	8.6	2 219	1 972
1998	6.8	2 370	2 106
1999	5.1	2 490	2 213
2000	5.4	2 625	2 333
2001	5.8	2 777	2 468
2002	9.1	3 030	2 692
2003	5.8	3 206	2 849
2004	1.4	3 250	2 889
2005	3.4	3 361	2 987
2006	4.6	3 516	3 124
2007	7.2	3 769	3 349
2008	11.48	4 203	3 736
2009	7.10	4 502	4 001
2010 (est 3rd quarter)	4.58	4 708	4 185

Based on the above assumptions, Table 26 shows the estimated direct treatment costs for high and low service level areas within the Olifants catchment.

Table 26: Estimated direct treatment costs for diarrhoea in the WMA

Description	< RDP (Low service level) Direct health costs	> RDP (High service level) Direct health costs
Estimated diarrhoea incidences	2 220 151	210 636
# of cases treated	(8%) 177 612	(5%) 10 532
Average Treatment costs	R3 769	R3 349
Total Direct Health costs	R669 419 899	R35 270 978
Total Direct health costs/million inhabitants	R301 million	R30 million

HIV/AIDS and the impact of this disease on the diarrhoeal disease and increase in diarrhoeal deaths have not been factored into these estimates. Based on current knowledge, diarrhoea deaths have largely increased from 6 350 in 1997 to almost 40 000 in 2006, mainly due to the devastating impact of HIV/AIDS (Groenewald *et al*., 2005; Steyn and Genthe, 2010). Also noteworthy is the fact that due to HIV/AIDS, diarrhoea deaths and therefore also diarrhoea morbidity have significantly increased in the adult population. This implies that the above cost estimates based on the method used by Pegram *et al.* (1998) and DWAF (2001) is rather conservative for the adult population.

Taking into consideration that these are conservative estimates, the above results clearly indicates that diarrhoea is a major health concern in the Olifants WMA. It also shows the importance of access to improved water, sanitation and hygiene and the associated possible cost saving associated with access to water infrastructure of above RDP standards. The people with access to above RDP water, sanitation and hygiene incurred five percent of the total cost compared to those with below RDP services.

8.2.4 Opportunity cost of Diarrhoea in the Olifants WMA

This section attempts to estimate the opportunity cost of water pollution in the Olifants WMA. Opportunity costs of water pollution are defined here as the foregone gross income as a result of morbidity and mortality incurred that is attributable to water pollution. Opportunity costs are estimated based on accurate measures of the BoD, which is calculated using a composite measure of the impact of a disease on the health of a population, known as the disability-adjusted life year (DALY) (Mathers *et al*., 2007; World Health Organization, 2010).

Waterborne diseases typically have two impacts on human health, namely morbidity (disease) and mortality (death). Both need to be quantified in order to determine sound value estimates of the environmental BoD. However, the health impacts associated with the environment have been overlooked in Africa, especially in terms of mortality (CSIR, 2010). The National Strategy for Managing the Water Quality Effects of Settlements (DWAF, 2001) mentions that poor water quality is a result of physical, social and institutional factors, and it explains in detail how these factors cause pollution. Within the context of environmental health, "environmental" pertains to "*all that which is external to the human host, including physical, biological, social, and cultural aspects, any or all of which can influence the health status of populations*" (Last, 2001). For example, physical breakdown of water supply and treatment infrastructure can result in poor water quality, which has a wide range of impacts on human health, social development and the environment (DWAF, 2001). In addition to health impacts, costs to the environment could potentially be significant in terms of non-use values such as bio-diversity and aesthetic degradation. Furthermore, other social costs such as loss of dignity are more often not quantifiable. Environmental and social costs that cannot be quantified should at least be investigated and discussed qualitatively but was not included in this study.

According to Pruss-Ustun *et al*. (2003), the harmful effects of the risk factor on human health and the extent and distribution of the effects must be known, in order to assess the disease burden attributable to a particular risk factor. Pruss-Ustun *et al*. (2003) remarks that an assessment of the environmental BoD can be used to raise awareness and strengthen institutional capacity for reducing the impact of environmental health risks on the population. A conventional BoD study quantifies the health impact at the level of the population, and could therefore form the basis of an environmental BoD assessment, in which the burden attributable to specific environmental risk factors is determined. Summary measures of population health make it possible to compare different estimates of the environmental BoD by standardizing the methodology; these estimates should be internally consistent and use an explicit, commonly-applied methodology (Pruss-Ustun *et al*., 2003). The National Strategy (DWAF, 2001) was only able to quantify the costs of treating diarrhoea, as well as downstream water treatment costs, as this was the only disease for which they could obtain reasonably verifiable statistics. Social impacts in terms of mortality were not accounted for (DWAF, 2001).

A number of methodologies have been used for the valuation of environment-related health and social impacts, particularly for estimating the morbidity (and to a lesser extent mortality) impacts of waterborne diseases. The most commonly applied approaches are the willingness-to-pay (WTP) approach (including the related concept of willingness to accept compensation, both of which can be estimated using the contingent valuation method (CVM)); the cost-of-illness (COI) approach; and the averting (defensive) behaviour approach; amongst others.

8.2.4.1 Methodological approach

This study attempts to estimate the BoD for waterborne diseases in the Olifants WMA attributable to water pollution, in terms of opportunity costs, based on the correlation between the environmental exposure (water pollution) and health (BoD).

- ***Research method***

Prevention and control of diseases require information about the causes of illness and of exposure to risk factors. Unfortunately, the assessment of the public-health importance of these has been hampered by the lack of common protocol and standardised methods to investigate the overall, worldwide burden of disease. However, the Global Burden of Disease (GBD) study by Murray and Lopez (1996) provides a standardised approach to epidemiological assessment and uses a standard unit, the disability adjusted life year (DALY). The DALY is an indicator of the overall BoD, combining a measure of both mortality (years of life lost due to premature death) and morbidity (years of 'healthy' life lost by virtue of being ill, measured by the number of years lived with the illness or 'disability'). Equation 1 shows the formula for calculating the DALY.

Equation 1: **DALY = years of life lost (YLL) + years lived with disability (YLD)**

The first value, YLL, is determined using the West model life-table (Coale and Guo, 1989) to determine age and sex-specific life expectancies; and the second value, YLD, is calculated on the basis of the incidence and duration of conditions resulting in non-fatal occurrences of disease, and is weighted according to the severity of the disability. In this way, the time lived with a disability is comparable to time lost due to premature mortality (relative to the life expectancy) (Schneider, 2001).

A third variable incorporated in the DALY is the value of time lived at different ages using an age-weight function. The middle age group, 9-54 years, is weighted more than the extremes (0-8 and 55+ years) (Schneider, 2001). This choice was based on a number of studies that have indicated that there is a broad social preference to value a year lived by a young adult more highly than a year lived by a young child, or lived at older ages (Murray and Lopez, 1996, Institute of Medicine, 1986). However, not all studies concur with the concept of giving the youth and the elderly relatively less weight as compared to the middle age group (Anand and Hanson, 1997). The project team agree with this social preference concept based on the stated preference of subjects investigated in previous studies

(Schneider, 2001). In addition, the project team is of the opinion that when using the BoD to determine the opportunity costs incurred by society in terms of gross domestic product (GDP) (or a similar economic indicator), assigning a similar weight for all ages is tantamount to double counting. The middle age group generally forms the bulk of the tax base within an economy and it assumes the responsibility for fending for the youth and elderly, hence their income is generally distributed along the entire demographic structure. Figure 17 shows the distribution of weights by age for all age groups as incorporated by DALY estimates.

Table 27: List of magisterial districts in study area *(Naudé et al., 2007)*

Province	**District name**	**Area (km^2)**	**Area in WMA (km^2)**
Gauteng	Ekurhuleni Metropolitan Municipality(EKU)	1924	24
	Nokeng tsa Taemane Local Municipality (GT461)	1968	963
	Kungwini Local Municipality(GT462)	2202	1778
	Lesedi Local Municipality(GT423)	1486	41
Mpumalanga	Albert Luthuli Local Municipality(MP301)	5559	16
	Msukaligwa Local Municipality(MP302)	6016	645
	Govan Mbeki Local Municipality(MP307)	2955	1152
	Delmas Local Municipality(MP311)	1568	1490
	Emalahleni Local Municipality(MP312)	2678	2678
	Steve Tshwete Local Municipality(MP313)	3976	3972
	Highlands Local Municipality(MP314)	4736	2025
	Thembisile Local Municipality(MP315)	2384	2384
	Dr JS Moroka Local Municipality(MP316)	1416	1400
	Thaba Chweu Local Municipality(MP321)	5719	3390
	Bushbuckridge Local Municipality(MP325)	2590	354
	KNP (MPDMA32)	11189	3144
Limpopo	Makhuduthamaga Local Municipality(LIM473)	2097	2097
	Fetakgomo Local Municipality(LIM474)	1107	1107
	Greater Marble Hall Local Municipality (LIM471)	1910	1910
	Elias Motsoaledi Local Municipality(LIM472)	3713	3713
	Greater Tubatse Local Municipality(LIM475)	4599	4599
	Greater Tzaneen Local Municipality(LIM333)	3243	776
	Ba-Phalaborwa Local Municipality(LIM334)	3004	1815
	Maruleng Local Municipality(LIM335)	3244	3244
	Polokwane Local Municipality(LIM354)	3766	1234
	Lepele-Nkumpi Local Municipality(LIM355)	3463	3360
	Mookgopong Local Municipality(LIM364)	4271	3231
	Modimolle Local Municipality(LIM365)	6228	47
	Bela-Bela Local Municipality(LIM366)	3376	1144
	Mogalakwena Local Municipality(LIM367)	6166	49
	KNP (LIMDMA33)	10118	778
	TOTAL	**118672**	**54562**

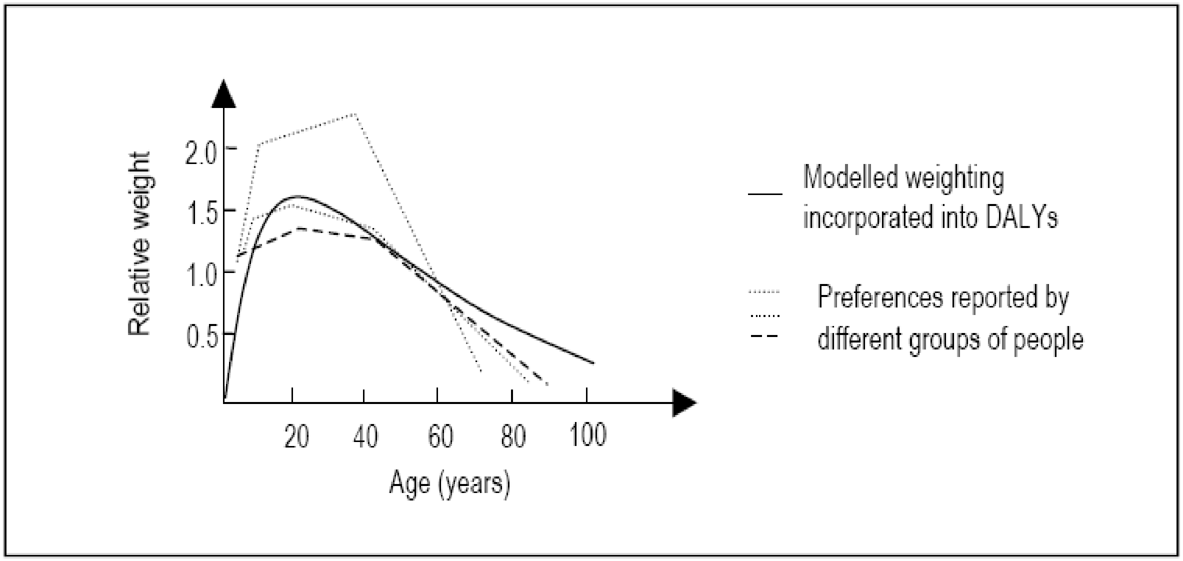

Figure 17: Relative weight of a year of life lived by age (Murray and Lopez, 1996)

The fourth variable incorporated in the DALY relates to time preference and involves the choice of a discount rate for anticipated future loss. Discounting implies a greater preference for the present as opposed to the future. This is particularly important in the context of cost-effectiveness and cost-benefit assessments where future costs are discounted (Schneider, 2001). In the DALY formula, future years of life lived are valued less than present years, i.e. with the recommended 3% discount rate, this implies that one life saved today will be worth more than five lives saved in 55 years time (Anand and Hanson, 1997).

Foege (1994) considers the DALY as one of the most important public health developments of the past century. He states that the DALY concept has the potential to revolutionise the way in which the impact of diseases is measured and presented. The DALY combines years of healthy life lost due to disability with those lost as a result of premature death and quantifies the difference between the actual health status of a given population and some "ideal" or reference status. DALYs have been calculated for over one hundred specific diseases for eight demographic regions worldwide (Schneider, 2001). It is also the metric that will be applied in this study to quantify the BoD for waterborne diseases attributable to water pollution in the Olifants WMA. The BoD in terms of DALYs will then be multiplied by the average gross per capita income per annum to obtain an estimate of the opportunity costs associated with waterborne diseases in the Olifants WMA.

- *Rationale for the method*

Although the DWA (DWAF, 2001) acknowledged that water-related diseases can be caused by many factors, including poor sanitation and health practices in the home, the risks are significantly higher in an environment with water supply and sanitation insecurity. As mentioned earlier the DALY will be used for the estimation of pollution opportunity costs. Anand and Hanson (1997), in a critical review of the DALY metric, hypothesized that a measure more appropriate than the DALY would need to account for the way in which individual and social resources can compensate for the level of disability experienced. Rightfully, the individual's actual loss of productivity will depend on both his/her uncompensated disability and the factors which affect his/her ability to cope with that disability. Such 'compensated' disability weights would depend, amongst others, on their personal income (ability to employ a person to assist with preparing meals, daily activities, and the like), and on the availability of social services to attenuate the condition such as amenities designed for the disabled. The project team concur with the motion by Anand and Hanson (1997) which states that factors which affect the ability of the inflicted to cope with the impact of disease, should be accounted for when assessing the overall BoD. In this regard, 'uncompensated' disability is the appropriate parameter in question because it constitutes an approximation of direct pollution impacts on public health. Unabated impacts provide an objective reflection of the opportunity cost of water pollution because 'compensated' disability weights are contingent upon the sufferer's income level and the availability of amenities for the disabled which sends a wrong signal about the problem. 'Compensated' disability weights also introduce the equity based problems of ability to pay differences between the rich and the poor which are more pronounced in a developing country context such as South Africa. It was attempted to attach an economic value to this parameter as it does not accord with intuition to determine a pollution impact value, and then lessen it on the grounds that measures are in place to compensate people suffering from pollution.

A simple analogy can be used to explain how ability to pay introduces bias to environmental valuation. The 'compensated' disability weights for a rich and a poor individual, are bound to differ, notwithstanding the fact that both individuals could have been exposed to the same level of environmental risk. A rich person is more able to abate a disease's impact than a poor person who is more likely to suffer the full impact. This is a grave source of bias (similar to the case with the CVM) in environmental resource valuation exercises.

In the same vein, the level of availability of amenities for the disabled could also be a source of bias to economic BoD assessment exercises. Two individuals exposed to the same degree of environmental risk, are likely to experience different levels of infliction if their amenity endowment levels are different. The individual endowed with amenities for the disabled undoubtedly suffers less than the less amenity endowed person. This calls for measures to regularize the unit of human productivity losses – the potential of any able bodied person to contribute towards the economy.

In congruity with moral judgement, this study assumes that the physical and mental capacity of any South African is the same, hence, as a proxy for the potential of any able bodied individual in South Africa to contribute towards the economy, the Statistics South Africa (2010b) quarterly employment statistics were used to eliminate environmental valuation bias associated with ability to pay. Furthermore, this study assumes that humans exposed to the environmental risk factor of microbial pollution face the unabated prognosis attributable to the waterborne ailment.

8.2.4.2 Data sources, calculations and limitations

A limitation of this study was the lack of site specific and complete data on the burden of waterborne diseases for the Olifants WMA, which implied some inference from provincial data. Several studies on environmental BoD have been conducted in South Africa, for instance Bradshaw *et al*. (2000a; 2000b; 2003; 2006). The initial intention of this study was to estimate opportunity costs associated with waterborne diseases specifically for the Loskop area, but this was simply not possible because of data limitations which permitted only a broader generalization to the Olifants WMA. In addition, data was only available for diarrhoea (most commonly associated with gastroenteritis). In line with the study by Bradshaw *et al*. (2003), the researchers acknowledge that there are certain marked differences in mortality and morbidity between the various population groups in South Africa. However, since available statistics do not provide data on population group or other demographic or socio-economic variables, this study, out of necessity, ignored variations in mortality and morbidity between sub-populations. It was also difficult to estimate the BoD variations between provinces because reliable estimates of the burden of waterborne diseases from the World Health Organisation (WHO, 2009) are only available at a country level, however, they had the merit of being representative of South Africa. There is a paucity of environmental BoD research at the local level in South Africa.

Population-wise, an area weighted, meso-scaled GAP[2]-based population estimate for the Olifants WMA was calculated to be 3.2 million people (Table 28) (Naudé *et al.*, 2007; Van Vuuren *et al.*, 2003). Similar to the BoD study by Bradshaw et al. (2003), age weighting (see Figure 17) and discounting values were incorporated in this study. These may not necessarily reflect the values that would be selected by the South African population. However, without population-based data on health state preferences, age weighting or discounting, it is not possible to adjust for these. It is recognised that a difficult step in estimating the BoD for most diseases is matching existing population data to severity categories that have specific weights assigned to them. The weights can be obtained from discussions with experts, but should preferably be derived from population health-state-preference surveys, which are not available in South Africa

Diarrrhoea is not a notifiable disease and therefore incidence rates are commonly underestimated. The COI approach based on previous studies (Pegram *et al.*, 1998) calculated incidence rates for varying levels of water supply and sanitation access for the population within the WMA (in Table 23). For example it was estimated that 2 223 426 people were receiving water supply and sanitation services below the RDP standard in 1998.

Table 28: Population of the study area *(Naudé et al., 2007)*

Province	District name	Estimated population
Mpumalanga	Albert Luthuli Local Municipality(MP301)	568
	Msukaligwa Local Municipality(MP302)	13537
	Govan Mbeki Local Municipality(MP307)	104890
	Delmas Local Municipality(MP311)	47967
	Emalahleni Local Municipality(MP312)	435217
	Steve Tshwete Local Municipality(MP313)	182277
	Highlands Local Municipality(MP314)	14042
	Thembisile Local Municipality(MP315)	278517
	Dr JS Moroka Local Municipality(MP316)	244155
	Thaba Chweu Local Municipality(MP321)	51887

[2] The South African GAP (geospatial analysis platform) was developed specifically to address the problem of spatially incompatible 'large area statistics' and other limitations associated with indicators and related maps that portray the geography of need, development and sustainability in terms of an absolute, container view of space (De Lange *et al*., 2009). The underlying mesoframe methodology – developed by the Council for Scientific and Industrial Research (CSIR) Built Environment unit (Naudé *et al*., 2007) – overcomes the problem of spatially incompatible 'large area statistics' by re-scaling and assembling a variety of census, satellite imagery and other data sources in terms of a common set of meso-scale analysis units. This consists of 25,000 irregularly shaped meso-zones (approximately 49 km^2 or 7 km by 7 km in size). The demarcation of mesozones was determined using various types of boundaries (political, economic and biophysical). In particular, they were demarcated so as to nest within important administrative and physiographic boundaries, and to be connected to a digital road network for South Africa. A primary consideration was that these boundaries should correspond with travel barriers such as rivers and 'breaklines' between sparsely and densely populated areas. The data rescaling methodology is similar to the method that was used to derive the 1 km resolution Global Landscan population database (developed by the USA's Oak Ridge National Laboratory), where large area census information was disaggregated on the basis of fine-grained remote sensing information (Bhaduri *et al*., 2007).

Province	District name	Estimated population
	Bushbuckridge Local Municipality(MP325)	69661
	KNP (MPDMA32)	828
Limpopo	Makhuduthamaga Local Municipality(LIM473)	262726
	Fetakgomo Local Municipality(LIM474)	112232
	Greater Marble Hall Local Municipality(LIM471)	124510
	Elias Motsoaledi Local Municipality(LIM472)	247488
	Greater Tubatse Local Municipality(LIM475)	343468
	Greater Tzaneen Local Municipality(LIM333)	83570
	Ba-Phalaborwa Local Municipality(LIM334)	76904
	Maruleng Local Municipality(LIM335)	95779
	Polokwane Local Municipality(LIM354)	184051
	Lepele-Nkumpi Local Municipality(LIM355)	234225
	Mookgopong Local Municipality(LIM364)	12723
	Modimolle Local Municipality(LIM365)	401
	Bela-Bela Local Municipality(LIM366)	18929
	Mogalakwena Local Municipality(LIM367)	2631
	KNP (LIMDMA33)	77
	TOTAL	**3243260**

'Labour force', (as used synonymously with economically active population) is described as all persons of working age (15-65 years) who are employed or unemployed but available for work (StatsSA, 2003). There is about 31% of the basin population that make up the economically inactive population, which refers to any person not in the labour force, such as housewives/homemakers, students and scholars, pensioners and retired people, and any other not seeking work during the reference period (StatsSA, 2003). Employment refers to an activity in which a person is engaged for pay, profit or family gain or a combination of the three (StatsSA, 2003). Such employment could be within the formal or the informal economy of South Africa. Of the total labour force in the Olifants WMA, 53% are employed in the formal economy which presents 14% of the basin population. No similar figure for employment in the informal economy currently exists. None the less, the focus of the study now turns towards the economic active proportion of the population, since this part of the population support the other age groups. Consequently the focus was on the diarrhoea incidence rate of the economic active proportion of the population (between 15 and 65 years of age) for the Olifants WMA which was estimated on 1 048 693 incidences of diarrhoea per year.

Gross income is the amount of income received before taxes. Statistics South Africa (StatsSA, 2010b) released the quarterly employment results for the quarter ended March 2010. The average monthly gross earnings levels are displayed in Table 29.

Table 29: Average monthly income of the formal non-agricultural sector of South Africa

Average monthly earnings	Feb 2009 (Rand)	Nov 2009 (Rand)	Feb 2010 (Rand)	% change between Nov ’09 and Feb ’10	% change between Feb ’09 and Feb ’10
Including bonuses and overtime payments	9614	11020	11195	1.6	16.4

Source: Statistics South Africa (StatsSA, 2010b)

According to Statistics South Africa (StatsSA, 2010b) the most recent average monthly gross income figure per capita in South Africa for formal non-agricultural sectors is R11 195. The International Labour Organization (ILO) (1973) in its resolution concerning wage statistics mentions that, in the view of special problems associated with the collection of data for the agriculture sector, a special programme of wages statistics should be drawn up for the agriculture sector. The ILO further notes that although the international definition of labour cost is also applicable to the agriculture sector, statistics of labour cost in the traditional subsector of agriculture would not be meaningful since hired labour constitutes only a minor part of the labour input. For these reasons, it should be noted that the Statistics South Africa gross earnings estimates do not cater for the agricultural sector.

The average gross earnings as presented in Table 29 is the unbiased figure that will be used during the environmental valuation exercise, as a proxy for the potential capacity of an able bodied individual to contribute towards the GDP of South Africa (be they employed or unemployed, rich or poor, endowed with abating amenities or not).

Although provincial income data for the formal sector in Limpopo (R3 102 on average per month) and Mpumalanga (R2 375 on average per month) is available (StatsSA, 2002b; StatsSA, 2004b; StatsSA, 2002a; StatsSA, 2007b; StatsSA, 2007a; StatsSA, 2004a; Department of Economic Development and Planning, 2009) it was decided to use national income figures based on the following arguments:

- Labour migration patterns in South Africa. The country has a long history of labour migration and it is almost certain that significant numbers of people from Limpopo and Mpumalanga will search employment in neighbouring Gauteng or any other province. Local provincial income figures are therefore not representative.
- Pollution in Olifants WMA can (and does) originate from upstream sources, likewise, Olifants pollution has downstream effects, which implies that the economic impact of pollution is not confined to the Olifants WMA only.
- Furthermore, it was argued that all humans have the same potential. The development of such potential is partly due to a function of the environment and this implies that suffering individuals have similar brain and strength potential, so they should be valued the same. Water pollution of equal intensity, albeit in different regions, therefore needs to have the same value; one cannot argue that the same amount and extent of pollution have different damaging in terms of human capacity impacts across different geographical areas.
- The impacts on the informal economy are most of the time excluded from these kinds of studies. While it is estimated that the cash value of trade in the informal economy (bartering) is estimated to be at least 9.5% of the value of GDP (all production within South Africa irrespective whether they are locally or foreign owned)(Saunders, 2005) or 28.4% of GNP (production value of all South African owned companies

irrespective of whether they are within South Africa or not)(Schneider, 2002), pollution will also impact on these values. Given that there is no reliable way of estimating such impact it is anticipated that the choice for a higher income (R11 195) would partially account for this limitation.

Table 30 shows the formula used to calculate the water pollution opportunity costs incurred in the study area, and the source of data used.

Table 30: Method of calculating the opportunity costs

f(x)	Data	Source	Metric calculated
	Opportunity costs		
=	DALYs associated with waterborne disease per 1000 individuals per year	(World Health Organization, 2009)	
x	Olifants WMA at risk population	(Van Vuuren *et al*., 2003; Naudé *et al*., 2007)	Olifants WMA DALYs
x	Average per capita gross income per annum	(StatsSA, 2010b)	Opportunity costs

The aim of health interventions is to minimise the number of DALYs, that is, to promote a longer and healthier life for people (Bradshaw *et al*., 2003). The WHO country profile of environmental BoD for South Africa (WHO, 2009) specifies the DALYs for South Africa due to water pollution in terms of gastroenteritis. Their estimates are based on comparative risk assessment, evidence synthesis and expert evaluation for regional exposure and WHO country health statistics.

Table 31 indicates that the DALYs attributable to water, sanitation and hygiene are 8.1 per thousand people per year, while the actual number of deaths ascribed to this risk factor is 12 300 per annum in South Africa.

The purpose of this study was to determine the opportunity costs due to water pollution, by measuring the foregone gross income as a result of morbidity and mortality associated with waterborne diseases. This study assumes that the greater contribution towards GDP derive from the economically active segment of the population, which complements the social time preference incorporated in the DALY metric. From this understanding, it should be noted that the DALY already assigns a higher weight to the economically active population than to the youth and elderly.

Table 31: Burden of disease for water, sanitation and hygiene per year (WHO, 2009)

Risk factor	**Exposure**	**Deaths/year**	**DALYs per 1000 per year**
Water, sanitation and hygiene (diarrhoea only)	Improved water: 88% Improved sanitation: 65%	12 300	8.1

8.2.4.3 Results and discussion

Based on this data, it was possible to determine the opportunity costs of water pollution in the Olifants WMA, in terms of the burden of waterborne diseases as measured by DALYs associated with incidences of diarrhoea in the WMA. Table 32 shows the results of the opportunity cost calculation.

Table 32: Calculating the opportunity costs

Data	Values
DALYs associated with diarrhoea per 1 000 per annum for SA	8.1
Incidence of diarrhoea in the economic active population of the Olifants WMA	1 048 693
Total DALYs per annum due to water borne disease	8 494
Average per capita gross income per month (Rands)	R11 195*12
Opportunity costs (Rands/annum)	**R1 141 million/year**

This study estimated the opportunity cost of water pollution (based on DALYs associated with diarrhoea in the Olifants WMA) to be approximately R1 141 million per year for Olifants WMA. This represents at least 0.046% of national nominal GDP, which has been estimated at R2 500 billion for 2010 (StatsSA, 2010a) or approximately 1% of the GDP in the Olifants WMA.

The opportunity costs of water pollution are, in essence, estimates of the value (the benefit) of preventing water pollution. This is because these costs are tantamount to the benefits that society stands to enjoy if water pollution is prevented. They represent the upper bound of the magnitude of investment that needs to be made in order to prevent morbidity and mortality that can be brought about by allowing water to get polluted.

For example, for the Olifants WMA, when evaluating the worth of different water treatment interventions, the primary criterion to select an intervention would be to accept an intervention whose overall costs do not exceed the shadow value of R1 141 million per year. In simple terms, the costs of a selected intervention should not exceed the benefits that will accrue to society as a result of the morbidity and mortality that will be prevented.

8.3 Cost of prevention

Since 84% of all diarrhea incidences in SA is attributable to water and sanitation (Lewin *et al*., 2007) the cost of provisioning of improved water and sanitation services could serve as an indication of the cost of prevention of this disease. This cost can in turn be used as a proxy for the cost of pollution prevention due to the direct relationship between microbial pollution and diarrhoea. The cost of improved sanitation services to households are summarized in Table 33.

Table 33: Sanitation technology options (DWAF, n.d.)

Technology option	Capital Cost	Maintenance Cost	Operating Cost
Ventilated Improved Pit Toilet	R3000-R4 500	R80-200 per year if emptied once in 5 years	Household cleansing materials, water for hand washing
Ventilated improved double pit toilet	R3500-R6 000	R35-R135 every 2 to 3 years	Household cleansing materials, water for hand washing
Aquaprivy toilet with soakaway	R3 200-R5 500	R195-R390 per annum	Water for initial filling and up to 20 litres per day for flushing as well as Household cleansing materials, water for hand washing
Septic tank and soakaway	R9 100-R11 050	R300-R600 for emptying every 3 years	Water for flushing 60 litres per day and Household cleansing materials, water for hand washing
Conventional waterborne sewerage	R8000-R15000 per household including on-site facilities and the sewer mains and sewerage treatment facilities	R250-R500 per household per annum for maintenance of sewers, pumps and treatment facilities	R300-R1000 per household per annum

Similarly, the costs of providing improved water supply (Muruvan, 2002) are summarized in Table 34.

Table 34: The typical cost of providing basic levels of water services per settlement type (Adapted from Muruvan, 2002).

Typical costs for a basic level of service by settlement type for water (per household)			
Settlement type	**Urban Core**	**Dense Settlements**	**Villages**
Service Level	Yard tank or yard tap	Communal standpipe within 200 m	Communal standpipe within 200 m
Typical level of water consumption (per household per month)	6 kℓ	2.5 kℓ	2.5 kℓ
Capital Cost	R3500.03	R2699.93	R3399.99
Monthly operating and maintenance cost	R20.01	R12.04	R15.94
Total cost per household	**R3 520.04**	**R2 711.97**	**R3 415.93**

8.4 Discussion and conclusions

Poor water quality is a result of physical, social and institutional factors. While it is recognised that waterborne diseases can be caused by many factors, the risks are significantly higher in an environment with polluted water. This understanding of the cause-effect relationship forms the starting point for any argument for preventing such pollution and thereby reducing the incidence of disease (and the associated costs to society).

The primary aim of environmental BoD estimates is to inform policy, which requires site specific, complete and reliable value estimates. However, studies that have investigated the environmental BoD in South Africa are few and far between, and even fewer studies have determined the associated social costs in economic terms.

The DALY metric, can potentially revolutionise the way in which health problems are approached and addressed (Foege, 1994), notwithstanding its shortfalls that have been highlighted by several researchers (Anand and Hanson, 1997). By and large, the mere fact that it is a metric that combines morbidity and mortality precludes the imprudence of not fully accounting for the impacts of disease and injury. Given the high disease burden experienced in South Africa, the root causes need to be targeted and eradicated in order to improve health. It is therefore important to identify and assess the underlying factors leading to disease in particular cases. In a few instances, the occurrence of an ailment can be attributable to a particular environmental risk factor, while in most it cannot. Because of the complex nature of the cause-effect relationship in many cases, more research needs to be conducted in the field of environmental health to gain a better insight into these relationships.

It is important to assess the specific burden attributable to selected risk factors so as to identify the issues that need to be addressed to reduce the environment-related BoD in South Africa. This should be augmented by the economic analysis of the social costs associated with this BoD, so that the cost effectiveness of interventions can be determined. BoD information is particularly useful because it can be used to predict the health gains that proposed interventions (including regulations) will bring to a population. Such information could be used to guide policies and strategies, both in the health sector and in the environmental sector; to monitor health risks; and to analyse the cost-effectiveness of interventions. For example, the information can highlight the contribution of major environmental risk factors to the total disease burden of a country or study population. Or, it can be used to estimate changes in the disease burden and avoidable disease burden, following interventions to reduce an environmental risk factor or to change behaviour.

The estimates and assumptions used in this assessment are stated explicitly. Although they should be interpreted with caution owing to the number of assumptions made, value estimates provide an order of magnitude indication of the extent of the problem and should be applied to usher a paradigm shift of environmental health policy approaches that proffer more effective and sensitive interventions.

The total direct health costs attributed to diarrhoea in the Olifants WMA is estimated at R301 million per million inhabitants with access to below RDP level of services. In comparison, the cost incurred by the population with above RDP level of services is R30 million per million inhabitants. Therefore, if the provisioning of RDP level sanitation and water services is considered to be a prevention measure for diarrhoea, the benefits in terms of direct health cost savings are tenfold.

In terms of the cost of preventing microbial pollution of water resources, improved sanitation services are required. The direct cost impacts of diarrhoea in areas with below RDP levels of water and sanitation services in the Olifants WMA is estimated at R669 million and the opportunity cost at R1 141 million per annum. The cost of conventional waterborne sewerage is between R8 000 and R15 000 per household including the on-site facilities and the sewer mains and sewerage treatment facilities and VIP toilet between R3 000 and R4 500. At an estimated household size of six individuals, 370 571 households require improved sanitation services at an estimated capital cost of between R1 112 million (VIP toilets at R3000) and R5 558 million (waterborne sanitation at R15 000).

Table 35: Cost comparison of microbial pollution (Cost of Diarrhoea) and prevention through improved sanitation systems to below RDP service households in the Olifants WMA.

Cost of illness	Provision of VIP toilets	Provision of waterborne sanitation
Direct cost: R669 million/annum	Capital Cost: R1 112 million (once off)	Capital Cost: R5 558 million (once off)
Opportunity cost: R1 141 million/annum	Operational cost: R29 million/annum	Operational Cost: R185 million/annum

Based on the summary provided in Table 35 it can be concluded that it will be cheaper to prevent pollution than to suffer the cost of illness related to diarrhoea in the study area. The capital cost of providing VIP toilets will be offset against the saving in direct cost of illness incurred in 1.6 years and that for waterborne sanitation in 8 years.

9 Cost of sedimentation

9.1 Sedimentation in the Olifants WMA

Sedimentation is a serious concern in the lower Olifants catchment with sedimentation levels in the Phalaborwa Barrage at 37.1% (Aquastat, 2008). The sediment production potential in the Olifants River is illustrated in Figure 18.

Downstream of the Flag Boshielo dam lies an area with little industry or infrastructure. However, it is also the area with the highest population and the fastest rate of population growth which results in poor agricultural practices based on short-term food supply priorities. This causes denuded grassland and increased erosion (DWAF 2004a). Overgrazing, stock density and stripping of vegetation for fuel in the middle Olifants catchment area must be considered as the biggest reason for the sediment pollution further downstream.

Suspended solids tend to accumulate with increasing distance down a river. The impacts of high TSS will therefore increase in severity in a downstream direction towards KNP, which is potentially one of the most vulnerable areas to increased pollution in this catchment. The Phalaborwa Barrage experiences operational problems due to the high levels of sediment. Operation of the sluice gates at the Phalaborwa Barrage to meet the ecological reserve flow requirements is extremely difficult. In addition, fish kills in the KNP have been associated with increasing levels of suspended solids (DWAF 2004a).

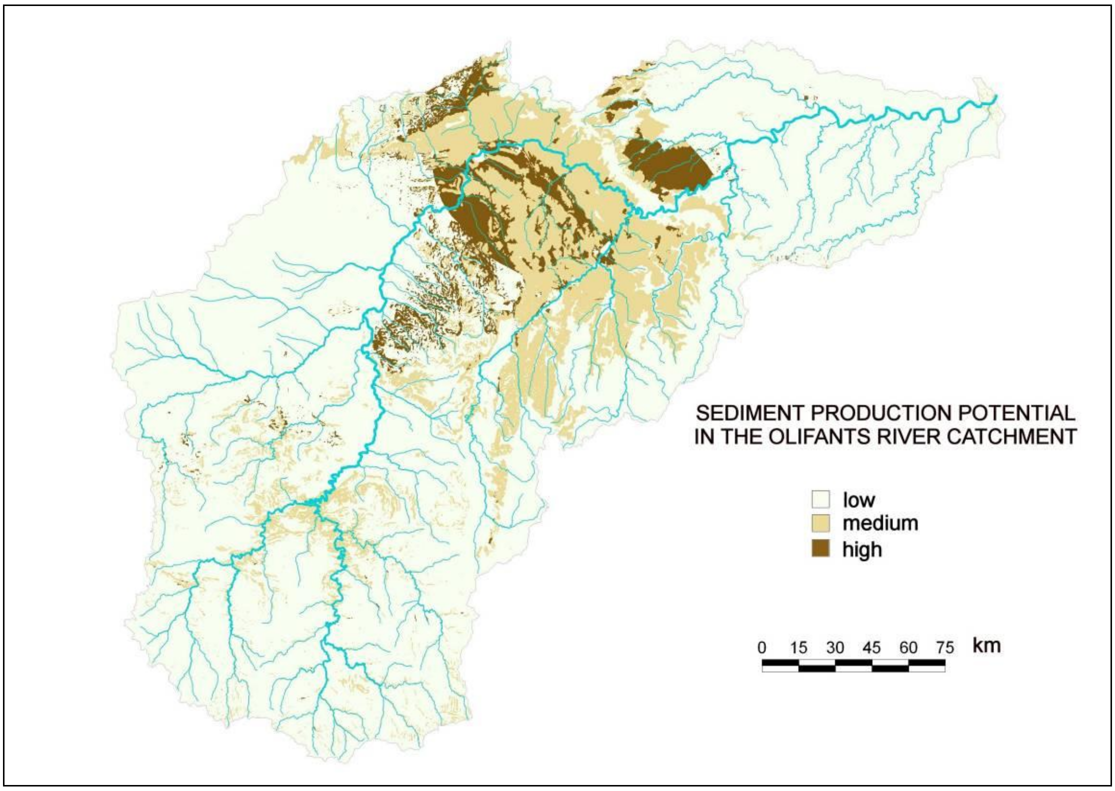

Figure 18: Map showing areas of low, medium and high sediment production potential in the Olifants River Catchment (Moolman *et al.*, 2004)

9.2 The cost of sedimentation in the Olifants WMA

Soil erosion is a common sight in South Africa as a result of poor land-use practices combined with erodible soils. Water has been identified as the main cause of soil erosion and silt rich run-off leads to the sedimentation of water resources (Le Roux *et al.,* 2007). The loss of fertile soil is a huge concern. The observed average sediment yield per unit area in South Africa varies between 10 to more than 1000 t/km^2/year (Braune and Looser, 1989). Eroded soil can also have various negative externalities including sedimentation of downstream reservoirs that causes reduction in storage capacity (Kapadia *et al*., 2002). Sedimentation damage to agricultural land resources through overwash of infertile material, impairment of natural drainage and swamping due to channel aggregation (Braune and Looser, 1989) is not easily determined. The economic benefits of dams are also reduced in terms of water supply, hydropower, recreational activities and flood control. It is therefore difficult to assess the full cost impact of sedimentation. In addition, erosion reduces soil productivity very gradually (Kapadia *et al.* 2009). This reduction in soil productivity requires additional input by farmers in terms of fertilizer and soil conditioners to maintain the crop yields over time, adding a cost burden that is not included in this study.

The main source of sediments in the Olifants WMA is opencast mining activities, deforestation and agriculture. Unfortunately, the crops that present the greatest erosion problem are those of considerable value either for industrial purposes or as food crop upon which the survival of the world's population depends (Kapadia *et al.,* 2002).

Soil conservation practices should include reforestation and a change from cropland to pasture where possible. Rotation methods involve strip cropping and mulching while soil management techniques use conservation tillage (Kapadia *et al.*, 2002). Mechanical methods of soil conservation include terracing and building structures to function as silt traps. Sediment traps and basins provide a cost effective means to reduce the amount of suspended solids in surface runoff (Ferreira and Waygood, 2009).

9.2.1 Methodology and data inputs

Two studies reported in literature, one by Braune and Looser in 1989 and the other by Sawadogo in 2008, set out to determine the cost impacts of sedimentation in South Africa. Both studies (Braune and Looser, 1989; Sawadogo, 2008) estimated the cost of sedimentation based on the total storage and mean loss of storage capacity for 170 reservoirs. The replacement value of the lost storage capacity is then used to determine the cost of sedimentation. The construction costs of new storage capacity can be calculated per cubic meter of storage volume (Braune and Looser, 1989).

Based on this approach, it was possible to estimate the cost to recreate the lost storage capacity at the Phalaborwa Barrage (37.1% = 2 096 150 m^3) as indicated in Table 36.

Table 36: Cost of dam construction in the Olifants WMA.

Project	Storage capacity created	Cost of project	Construction cost/ m^3	Reference
Raising of Flag Boshielo Dam	88 million m^3	R200 million	R2.27	RSA, 2003
Construction of De Hoop Dam	347 million m^3	R2.5 billion	R7.21	DWA, 2010a
Recreating lost capacity at Phalaborwa Barrage	2 096 150 m^3	R4.7 million R15 million	R2.27 R7.21	

The raising of the Flag Boshielo Dam in the Olifants WMA by five meter was done to increase the storage capacity from 100 million m^3 to 188 million m^3. The project was complete in 2006 (RSA, 2003). Construction of the new De Hoop Dam in the Steelpoort valley started in July 2007 and will create 347 million m^3 of storage capacity (DWA, 2010a).

Alternatively, lost storage can be recovered using sediment removal techniques such as mechanical or hydraulic dredging and flushing, but these approaches are usually quite expensive (Kapadia *et al*, 2002). Basson and Rooseboom (1999) suggest that the same yield benefit can be realised by dredging a much smaller volume compared to a raised dam with additional storage created. However, the cost of dredging is generally believed to be higher per unit volume storage than that of raising a dam wall or constructing a new one (Basson and Rooseboom, 1999).

9.2.2 Results and discussion

The calculated cost of the loss in storage capacity in the Olifants WMA equates to a value of between R4.7 and R15 million. Replacement of lost storage can be achieved by construction of new storage through raising the dam wall or construction of a dam at a new site and by removal of the sediments. However, the new storage option has become very limited due to a lack of suitable sites for new dam construction. If reduced accumulation or removal of sediment is technically possible, its economic viability is likely to depend on physical, hydrological and financial parameters (Palmieri *et al.*, 2001). The results of a study by Palmieri *et al.* (2001) show that "sustainable management of reservoirs is economically more desirable than the prevailing practice of forcing a finite reservoir life through excessive sediment accumulation".

Capital treatment costs of water with high turbidity is increased by increased capacity requirements of the works and special design requirements such as pre-sedimentation tanks and special sediment removal equipment. Operational costs are increased due to increased flocculent and disinfectant requirements. The disposal of the sediment is another major problem and a huge cost impact. The annual increment on capital outlay on water treatment due to higher than normal turbidities was estimated at 2% of the total capital cost (Braune and Looser, 1989). The total off-site cost impact of sediments in South Africa in 1989, excluding the environmental damage, was estimated at R90 million per year (Braune and Looser, 1989).

Storage lifetimes of reservoirs can be increased using different desilting techniques. Sediment sluicing and flushing are the most common techniques used (Brandt, 2000). Management methods that can be applied to prevent sediment pollution of water resources (Palmieri *et al*., 2001) include:

- Reduction of sediment yield by measures in the catchment area
    - Soil conservation measures
    - Correction of land-slides and accelerated erosions and reforestations or
    - debris dams, intercepting coarse grained sediments, in mountainous streams.
- Sediment routing[3] through construction of off stream reservoirs, construction of sediment exclusion structures and by sediment passing through the reservoir (e.g. sluicing).
- Sediment flushing, whereby the flow velocities in a reservoir are increased to such an extent that deposited sediments are re-mobilised and transport through bottom outlets; and
- Sediment removal by mechanical dredging or by hydraulic removal (siphoning).

9.3 Cost of sediment control measures

The costs of sediment control measures are dependent on a number of factors:

- Volume of water passing through the trap or settling facility
- Rainfall recurrence interval that the trap or facility must accommodate
- Velocity of the water passing through
- Silt load
- Particle size distribution
- Access for maintenance purposes.

The initial capital cost of a large settling facility will be high while the use of conventional silt traps is usually more economical. The operating costs of the silt traps are high as regular maintenance is required while the operating costs of large settling facilities are relatively low.

[3] Routing is an operation that allows sediments to pass through a facility without deposition

9.4 Conclusion

At an average storage loss rate of 0.35% per annum, the total storage loss per annum for South Africa was calculated at 105 Mm^3 at an estimated cost of R53 million per annum (Braune and Looser, 1989). A more recent study by Sawadogo (2008) estimated the storage loss rate at 0.28% per annum (109 Mm^3). At an estimated construction cost of R12/m^3, the annual loss of capacity is R1 300 million, excluding indirect costs (Sawadogo, 2008).

The direct cost of sedimentation calculated as a loss in storage capacity in the Olifants WMA is between R4.7 and R15 million. It was not possible to calculate the costs of sediment prevention for comparative purposes. However, loss of storage capacity in dams is a serious concern for the country as whole as it may impact on water security over time.

High TSS levels will however also impact on water abstraction facilities (for agriculture and potable supply) in the area. Major upgrading of such facilities will eventually become necessary to cope with the likely increase in sediment loads as well as the associated organic and inorganic pollutants. In 1995, an upgrade project of the Olifantspoort water treatment plant included improvements to the chemical coagulation and sediment storage dams, at a total cost of more than R17 million (DWAF 2004a). The adsorption of heavy metals and organic pollutants to sediment particles may increase the removal of these pollutants from the water together with the removal of the suspended solids. However, the increased sediment loads causes operational concerns at the treatment works, including blocking of filters, the requirement for more frequent desludging and increased volumes of sludge to dispose of. Toxicity of the sediment may qualify the sludge as a hazardous material, vulnerable to leaching. Although no specific data is available regarding the potential toxicity of the sediment in the rivers of the Olifants catchment area, such information from other rivers in industrialised areas indicate that sediment toxicity (i.e. hydrocarbons) could reach dangerous levels in the future (Maljevii and Balac, 2007).

The design and construction of man-made structures such as dams and weirs should therefore consider the impacts of sediments to optimise their operational lifespan (Rooseboom, 2002). Increased TSS levels associated with increased mining and agriculture will therefore require changes to current designs and treatment processes at both potable and irrigation water treatment facilities, resulting in increased cost of water treatment.

A sediment management plan is required for the Barrage to achieve the required downstream flow patterns at an acceptable water quality regarding suspended solids, including periods when the sluice gates are opened to allow sediment scouring (DWAF 2004a).

10 Pollution Prevention

The cost of removing pollutants increases exponentially, which provides the incentive to rather prevent pollution than to clean-up after the pollution has occurred. Beyond a certain point, the clean-up costs exceed the harmful costs of pollution. Damaging chemicals that either cannot be degraded by natural processes or which naturally degrade very slowly, should not be released into the environment, or they should be released only in small quantities and regulated by special permits (Tyler Miller, 1996).

Long-term trends in pollution may result in a shift from salinity (mining related) dominated pollution problems to bacteriological (sewage) pollution problems (RSA, 2007). Mines are closing down resulting in the uncontrolled rising of underground water levels while malfunctioning sewage works at local authorities are an increasing problem impacting on downstream users in terms of treatment costs and "fitness for use" of the water (RSA, 2007)

10.1 Strategies to prevent eutrophication

10.1.1 Effluent standards

An effluent phosphate standard of 1 mg/ℓ ortho-phostphate (introduced in 1984) was the first step taken by the DWA in an attempt to control eutrophication in water bodies in sensitive catchments. The phosphate standard was therefore only implemented in certain identified "sensitive" catchments. This decision however did not consider the best available information on phosphorous removal technologies and the consequences of increasing P loads to water supply reservoirs. In addition, failure to comply with this standard has resulted in the progressively worsening problems experienced currently and into the future. Phosphorous is considered to be the most manageable of the nutrients that contribute to eutrophication of water resources and cost effective technologies for removal of phosphorous from wastewater are generally available (Hohls *et al.*, 1998). Despite the introduction of the special phosphate standard, in the Hartbeespoort Dam case, the phosphate levels in the dam remained unchanged (WRC, 2004) as a result of the high phosphate levels trapped in and released from the sediments. These findings clearly suggest that focusing on phosphates alone will not solve the eutrophication problem in South Africa. The so-called "eutrophication problem" in South Africa requires a multiple set of solutions, the most important of which is control of phosphorus loads and concentrations entering rivers and impoundments. Other, lesser, solutions include in-lake treatment. In order to control eutrophication, the source of phosphorus has to be reduced dramatically.

10.1.2 Introduction of zero-phosphate detergents

Limiting the use of phosphates in detergents is a popular approach to nutrient reduction in water resources worldwide (Quayle *et al.*, 2010). It is reported that this approach resulted in at least partial remediation of eutrophication in water resources (Quayle *et al.*, 2010). It is estimated that up to 35% of dam total phosphate loading could be eliminated through the removal of detergent phosphorous resulting in an estimated reduction in algal growth of up to 30% (Quayle *et al.*, 2010).

10.1.3 Diffuse source controls

The impact from agricultural run-off could be reduced by the introduction of more efficient use of agrochemicals combined with increased efficiency of irrigation systems.

Urban run-off is another diffuse source requiring urgent attention especially grey water management in these areas needs to be addressed. Provisioning of basic water services through yard taps without proper sanitation systems is a case in point. Improved sanitation systems that are properly operated and maintained will significantly reduce diffuse sources of nutrients from informal settlements.

10.2 Strategies to prevent salinisation

10.2.1 Mining

There are several reasons for pollution generation in the mining operations leading to AMD formation and salinisation of water resources including:

- The dewatering of ground water to allow mines to operate. This dewatering only occurs during the mining operations, but impacts on surface water.
- Decanting of underground mine water after mine closure, which will also impact on the surface water. This way of pollution can continue for many years to come.
- Ingress of ground water into the underground mining operations.
- Rainwater infiltration through tailing dams, affecting surface and groundwater.

Effective pollution prevention will therefore be achieved through a number of approaches to minimise water ingress during the various stages of planning, development, operation and closure of a mine. Pollution prevention measures should therefore include (Pulles *et al,* 2005):

- The application of water management measures e.g. seals, water diversion away from ‘hotspots’ within the underground operations that are aimed at minimizing the potential for water quality deterioration due to the oxidation of sulfide minerals.
- Locate waste residue deposits in areas where there is a minimum potential for contamination of the ground and surface water resource and construct water management facilities to intercept and contain any contaminated runoff and/or seepage.
- Apply appropriate geochemical assessment techniques to the design of mine closure strategies for underground mines in order to identify those options that will minimize the long-term pollution risks of such facilities.

All mines must now be able to demonstrate that reasonable measures have been taken to prevent clean water from becoming polluted. Therefore, the fundamental principle is to prevent, inhibit, retard or stop the hydrological, chemical, microbiological, radioactive or thermodynamic processes that result in contamination of the water environment at the point of contact, or to implement physical measures to prevent or retard the migration of the generated contaminants to the water resource (Pulles, 2006). Coal mines have adopted a water management policy that aims to maximise clean and dirty water separation, minimise the import of clean water, maximise the re-use of dirty water and the treatment of dirty water.

10.2.2 Agriculture

The most significant pollution prevention measure to reduce salinity emanating from agriculture would be a change in irrigation practices to reduce run-off from agricultural land. It is expected that practices of over irrigation will partly be addressed through the introduction of water use charges.

10.2.3 Industry

Historically, the focus for treatment of effluents has been on end-of-pipe treatment technologies. However, current trends are moving towards waste minimization and pollution prevention which can be implemented in a variety of ways including:

- Process changes to prevent waste generation (cleaner production)
- Water conservation strategies could be implemented to reduce the volumes of effluents produced
- Separation and recovery of economical valuable components from waste water streams
- Process changes to reduce the sludge volumes and long-term liabilities issues.
- Separation of toxic and non-toxic sludge during the neutralisation of acidic effluents

10.3 Strategies to prevent microbial pollution

Waterborne sewerage systems are often viewed as the preferred sanitation option; however, failing waterborne sewerage systems are increasingly causing pollution impacts, both in terms of microbial pollution and eutrophication of most river systems in South Africa. Investments in maintenance and repairs need to be seen as vital to the sustainability of sanitation service delivery in the medium to longer term. Due to neglect, many local municipalities have built up a backlog in maintenance, refurbishment, renewal and replacement orders (Oelofse *et al*., 2008).

In addition to the getting existing WWTW to operate at acceptable standards, areas where sanitation backlogs exist must be targeted for the provision of at least basic levels of sanitation services. Within the context of sustainability, efforts must be made to avoid, stop or at least minimize environmental degradation. Enforcement of environmental legislation is therefore a key factor to prevention of pollution of our water resources (RSA, 2007).

10.4 Strategies to prevent sedimentation

Sedimentation often results from poor land-use practices including non-contour ploughing, deforestation and over grazing. Soil and water conservation programmes can therefore significantly reduce sediment loads entering the water resources. Sediment loads entering the dams can be minimized through upstream trapping of sediment (debri dams or vegetation screens) or bypassing of high sediment loads (Sawadago, 2008).

11 Discussion and Conclusions

11.1 Discussion

The intentions of this project were to, as best possible, estimate and compare the costs (in a broader sense) of water pollution to that of water pollution prevention regime. However, severe data limitations have limited the number and type of comparisons which could be included to illustrate this problem. Consequently the study included cost estimations for the following:

- Direct cost of waste water treatment at WWTWs in the upper Crocodile catchment.
- Direct cost of water purification at Rietvlei WPW.
- Income losses in the Loskop WUA due to salinisation of irrigation water.
- The direct cost of microbial pollution (diarrhoea) in the Olifants WMA.
- The opportunity cost of microbial pollution (diarrhoea) in the Olifants WMA.
- The capital and operational costs of improving access to above RDP levels sanitation services in the Olifants WMA.
- Sedimentation as the loss in storage capacity in the Olifants WMA.

The comparison between waste water treatment (pollution prevention measure) and water purification to potable standards has indicated that capital and operational costs associated with both types of facilities are in the same order of magnitude. It was found that economies of scale have significant impacts on unit cost pricing (larger facilities reap the rewards of economies of scale compared to their smaller counterparts). It is also clear that the technology employed for the purification of drinking water from surface water sources is becoming more sophisticated. This tendency is clearly due to ongoing pollution of these surface water resources and could imply a significant increase in the cost of drinking water in the near future.

The cost of pollution prevention measures required to curb pollution from its sources is not easy to calculate. However, the cost effects of water pollution on society are clear. Farmers in the Olifants River catchment can suffer losses due to increased salinity of between R30 107 (maize) to R703 495 (citrus) annually per typical farm unit (25hectares) in the WUA. Thus, considering this typical farm size and given that the irrigation area in question is 16 136 ha, there are in the order of 645 farms which implies an estimated total loss in gross margin above specified costs as a result of increased salinity in the Loskop WUA of between R19 million and R454 million per annum. Considering the fact that water abstracted for irrigation purposes is generally not treated before use, it is of the utmost importance to protect the quality of irrigation water through pollution prevention measures. It is not only required for the protection of the farmers and their interest, but in the interest of food security in the country.

The cost impact of microbial water pollution on human health is another cause for concern. Areas in the Olifants WMA with low levels of water and sanitation services suffer direct health costs of an estimated R699 million/annum due to diarrhoea. This is cost is tenfold higher when compared to the cost of illness of people with access to above RDP levels of water and sanitation services (R301 million/million inhabitants below RDP vs. R30 million/million inhabitants above RDP).

The direct cost of sedimentation calculated as a loss of storage capacity in the Olifants River catchment to date amounts to between R4.7 and R15 million. Loss of storage capacity can in the long run impact on water security in certain areas in this catchment. It is however significant to note that the annual loss of storage capacity for the country amounts to R1 300 million, excluding indirect cost (Sawadago, 2008).

11.2 Conclusions

Returning to the hypothesis of the project and an argument which prefer a pollution prevention regime to a pollution clean-up regime, it is clear that water pollution impose a significant burden on society in and outside the study areas. This project has put some effort in the quantification and valuation of such a burden. Numerous challenges hinders a fully inclusive and accurate estimate, however, the absolute quantification of the burden is not necessary as the central point is clear: water pollution poses a serious threat to human health in the study area. One could debate possible reasons for the current poor state of the systems under investigation, but such a debate will not necessarily bring to the fore a workable solution to the problem and for as long as such a debate continues, those incapable of taking remedial or self-protective action will continue to bear the burden of pollution.

It is imperative that the improvement of water and sanitation services to above RDP levels is prioritized. Furthermore, in addition to improve service delivery, awareness of hygiene and education on the correct use of the sanitation systems should also be prioritised.

Pollution is by nature a social burden where the burden of proof of traceability towards the responsible polluter fell on those who bear the pollution burden. This is unfortunate because power relations play an integral role in this situation and therefore present a situation where government action is required to engage the problem (few privately owned entities have the political and financial power to take the lead in this situation). The argument therefore returns to two main requirements to alleviate the pollution problem: first, pollution prevention and cleaner production technologies needs to be adopted to decrease the pollution load (at source) on the system, and secondly, improved management of treatment facilities is of critical importance to improve treatment standards and to decrease pollution levels. Pollution prevention prevents externalities and is thus considered a “better and cheaper” strategy as compared to carrying the cost burden of pollution. Pollution prevention costs are small (but direct) in comparison to that of the burden to society (an externality) including commercial agriculture affected by polluted water.

It is also argued that the current waste water treatment technology can deal with the water pollution issues covered in this study, if operated to acceptable standards. The upgrading, maintenance and proper operation of these facilities are of higher importance than the development of new technologies. However, in the case of WPW whose product (potable water quality) suffers from the current levels of pollution in the feed water, technology upgrades are essential.

12 Recommendations

Based on the fact that this report concluded that the costs of treatment at the point of use are likely to increase with associated decreased quality of intake water the following recommendations and way forward are recommended:

- **Prevention is better (and cheaper) than cure**

The report recognized the fact that the water resources in both the case study areas are polluted. While some of the pollution are and have been assimilated by the water resources in these areas, the water quality is deteriorating over time and distance and might soon be polluted beyond the point of return for treatment to ensure fitness for use.

Apart from the fact that the water quality is deteriorating, the cost of treating the water to improve the quality for the intended use is also increasing. Soon, it will be too expensive to treat the water for its intended use, which will have various water security implications.

- **Prioritise improved access to water supply, sanitation and hygiene**

Diarrhoea associated with a lack of access to safe drinking water is a concern in the Olifants Water Management Area and should be addressed. Although hypothetical diarrhoea incidences were calculated based on actual population figures, actual numbers of people with access to water supply infrastructure, and reported diarrhoeal incidence rates and deaths, this section highlighted the need for access to improved water supply, sanitation and hygiene. The diarrhoea morbidity and resultant direct treatment and transport costs in the Olifants Water Management Area could be drastically decreased if more people gained access to improved water supply, sanitation and hygiene. Not only will the chance of infection be reduced, but further costs can be saved as the severity of disease would decrease, resulting in reduced treatment requirements and associated costs.

- **Implement and police the 'Polluter Pay Principle'**

The external cost associated with water pollution is costing the country billions per annum while water security is also at stake. The implementation of the polluter pays principle also requires better policing

- **Implementation of the WDCS should be prioritised**

A water pollution audit on WMA level is proposed to form the baseline for the enforcement of the "polluter-pays principle" as a partial solution. Implementation of the WDCS should be prioritised to ensure that cost of pollution is internalized.

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Appendix 1: Loskop WUA profile

Contact person	Dirk Ferreira
Postal address	PrivateBag X 8684, Groblersdal, 0470
E-mail	djferreira@ctecg.co.za
Tel	0877503820 / 0829271219
From where do you receive the bulk of your water? (river, bulk storage dam or bulk supply system and/or service provider?)	Bulk storage dam
What type of distribution system do you operate? (open or lined canals; pipelines or direct pump ext.)	Concrete Lined canals , pipelines
Your rainy season?	Summer
When is your peak demand? (months)	Aug – Jan
When is your peak inflows? (months)	Dec – Mrt
Financial year (month)	Apr – Mrt
Government scheme? (yes/no)	No
Total enlisted hectares in your system	16136 ha
Average water allocation (m3/ha/year)	Depending on availability but capped on 7700 cubes per hectare per year
Please provide me with a summary of your user breakdown in terms of enlisted hectares (e.g. 600ha to 50 commercial farms; 20 to municipality)	16136ha to 355 commercial farmers There are also 8 industrial users in the system (not included in above mentioned hectares).
Your main commercial irrigated crops? (type and hectares)	Cash crops: Wheat, Potatoes, Maize, Sweetcorn, Peas, Tobacco, Soy, Cabbage, cauliflower, Pumpkin, Cotton, Permanent crops: Citrus, Grapes, Macadamia, olives
Your approximate volume of sales per year (Mm^3/year)	?
Water tariff system? c/m^3	14c/m3
What is the average price for water use rights in your region? (R/ha)	92.35 R/ha
Is water quality a problem in your system? (If yes, please explain why.)	Yes, the nitrates is high and algae is also a problem. Salinity is always a problem.
Main pollutants?	Upstream municipalities
Can you provide us with some time series data on these pollutants?	No not yet – busy with research (WRC project - CSIR involved)
Do you have data on the impact of polluted water and crop yield?	no
Notes/Comments	Call me if you need more info!

Appendix 2: Summary of standardized crop budget data as employed in this study

	Maize	Wheat	Potatoes	Citrus
Market price (R/kg)	1.53	1.89	1.95	1.65
yield (kg/ha)	7871	5466	41 045	45000
Typical farm ha (eg 20ha farm)	25	25	25	25
Total variable cost (R/ha)	9234	6985	48405	35840
Fixed cost including irrigation (R/ha)	1197	474	14849	25247
volume of water used to realise the yield (cubes/ha)	5930	7400	5650	10510
water constraint (cubes/ha)	5930	7400	5650	7700
Total water available (cubes)	148250	185000	141250	262750

Appendix 3: SAPWAT simulation run – Details

	Maize	Wheat	Potatoes	Citrus
Weather station	Loskop-Proefstas	Loskop-Proefstas	Loskop-Proefstas	Loskop-Proefstas
Option	medium grower early plant; planting in October; centre pivot irrigation	plant05/25; planting in may; centre pivot irrigation	standard; planting in August; centre pivot irrigation	average; average; planting in June
Climatical region	semi-arid with warm summers	semi-arid with warm summers	semi-arid with warm summers	semi-arid with warm summers
Total water requirement (mm/hectare)	1203	884	813	1732
Total irrigation requirement (mm/hectare)	593	740	565	1051