DEVELOPMENT OF A GENERIC MODEL TO ASSESS THE COSTS ASSOCIATED WITH EUTROPHICATION

Report to the
Water Research Commission

by

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EXECUTIVE SUMMARY

Eutrophication, which is the enrichment of the water environment with plant nutrients, causes excessive growth of phytoplankton (free floating algae) and rooted macrophytes. The presence of large numbers of phytoplankton in water bodies diminishes the quality of water resources for many users and costly treatment is often required to overcome its negative effects. The costs associated with eutrophication need to be estimated in order to consider and justify, often expensive, preventative and ameliorative measures. The overall objective of the project was to develop a generic model to assess the economic costs associated with eutrophication in South Africa and to apply it to the Vaal River system. The quantification of the costs associated with eutrophication will assist researchers and policy makers in identifying appropriate policies towards addressing eutrophication problems in South African river systems and will inform river catchment management decisions to remedy eutrophication problems.

The project was initially set out as part of a PhD study. However, due to changing circumstances with personnel the project was re-organised and conducted as several MSc projects. This report is in large part a summary and discussion of these MSc projects and additional relevant literature within the scope of the project. The MSc theses are available on the CD accompanying this report. The economic sector costs of eutrophication considered within this study are indicated in Figure 1. Due to capacity constraints the policy response costs and biodiversity costs of eutrophication were not investigated. The costs of eutrophication to the industrial sector were estimated indirectly by transferring the costs incurred by water supply services (water treatment costs) back to industries through the tariffs they pay for the water consumed.

Owing to the separation of the original study into several theses, the project shifted from an initially integrated evaluation of the total costs of eutrophication to a more statistical approach of estimating the individual sector costs associated with eutrophication. To conduct the econometric analyses, the typically daily observations of eutrophication indicators were transformed to annual averages effectively smoothing out the variability and peaks within the eutrophication data. However, the ecological and economic consequences of eutrophication are most often experienced as a result of changes in the level of eutrophication and the variability thereof at the time the variance occurs. For example, potable water treatment plants are typically able to cater for and treat the results of eutrophication occurring in the abstracted water. However, during times of severe eutrophication the algae present may produce high concentrations of taste and odour compounds and/or algal toxins, which can have a significant impact on water treatment costs during this period. Due to the averaging of the eutrophication data, the estimated relationships between eutrophication and the various economic sectors were relatively weak and the models were unable to capture the complete economic costs of eutrophication. An additional discussion was added to the study to further develop the concept of eutrophication costs and to identify how the costs estimated by the statistical models feature in terms of the total economic impact of eutrophication.

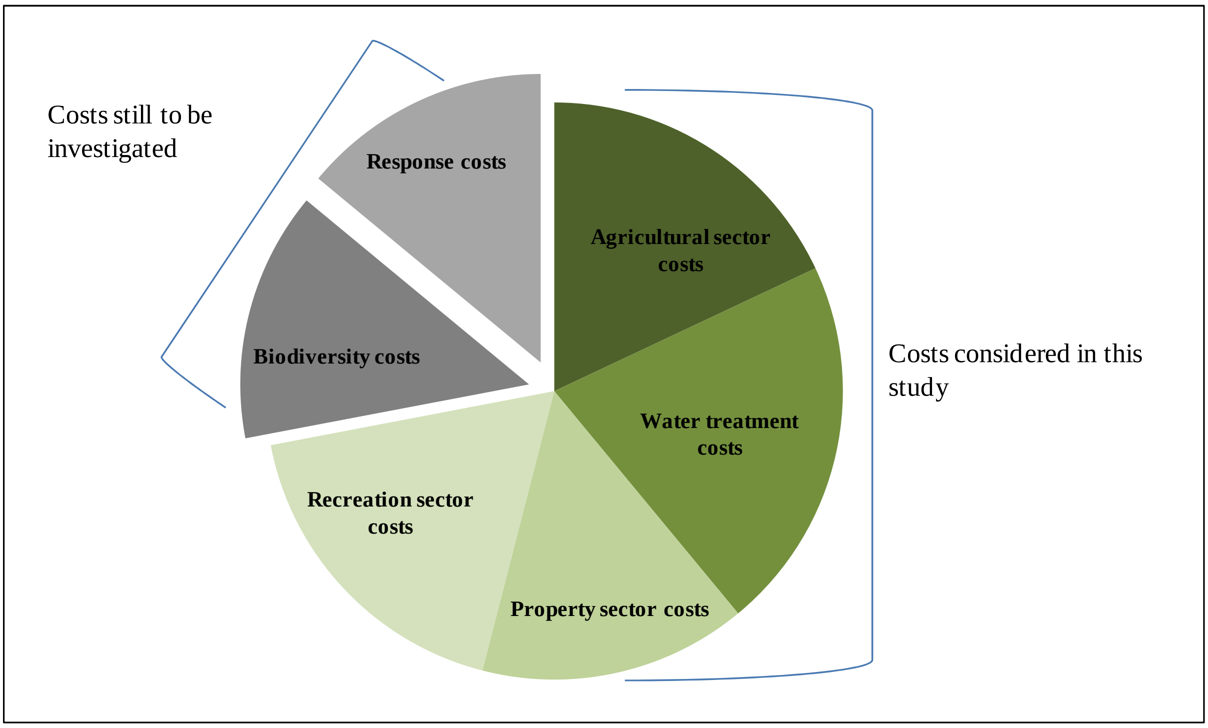

Figure 1: A representation of the costs of eutrophication modelled in this study and those costs still requiring investigation.

A thorough review of the literature was carried out to identify the impacts of eutrophication. The review described the characterisation of costs associated with eutrophication, Figure 2, and specifically identified the following as social damage costs:

- human health and livestock watering costs (costs to agriculture),
- water treatment costs,
- reduced value of water side property
- reduced recreational and amenity value of water bodies, and

- clean-up costs of waterways.

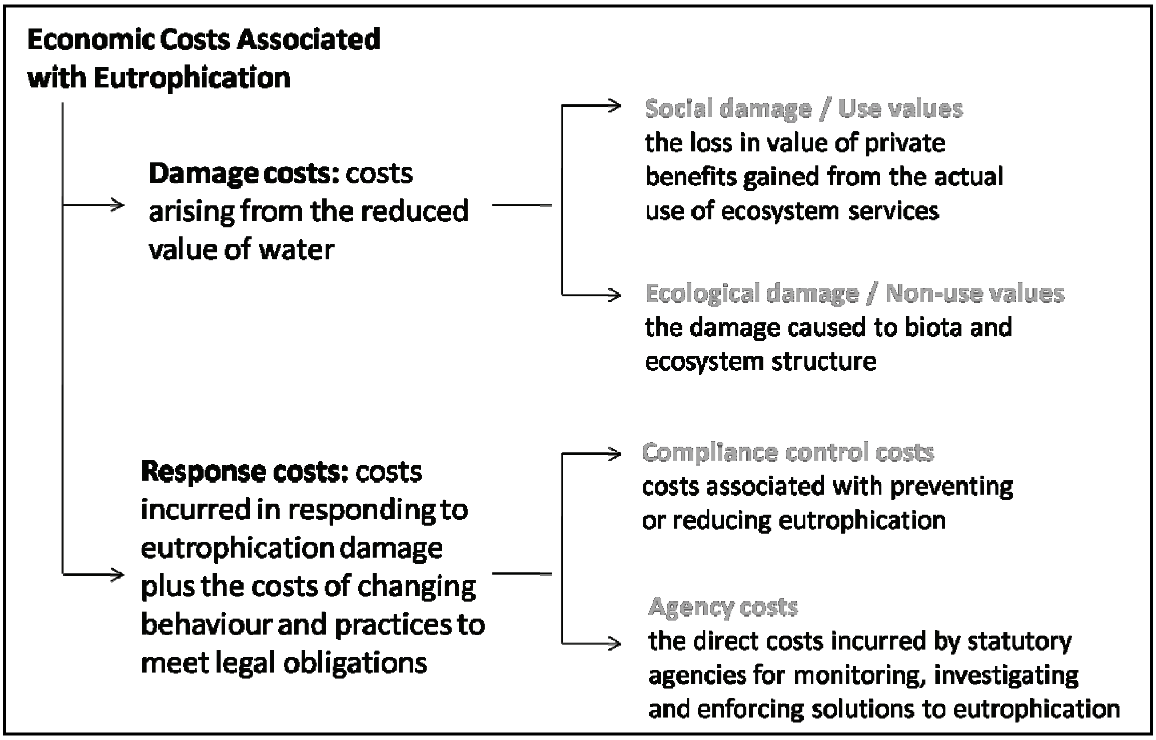

Figure 2: Categorisation framework of the economic costs associated with eutrophication.

Source: Adapted from Pretty et al. (2002)

Various methods of estimating the identified costs were reviewed. The methodologies applied in this study are described below.

Model description and results

The econometric models were specified for and applied to the Vaal River system, which was separated into three areas: the Grootdraai Dam situated in the upper area of the Vaal River, the Vaal Dam (middle Vaal), and Bloemhof Dam situated in the lower area of the Vaal River. The Vaal River is a major river system in South Africa and the primary supplier of water to the economic heartland of South Africa. The principal uses of water from the Vaal River system are domestic (including agriculture) and industrial (including mining and for cooling electricity power stations).

Regression analysis was used in the estimation of several of the cost models. Regression analysis is used to estimate the relationship between a dependent variable and one or more explanatory variables with the aim of predicting the average value of the dependent variable based on known values of the explanatory variables. For each economic sector considered in the study, the dependent variable was the relevant sector's total costs (e.g. total agricultural costs) and the explanatory variables were those variables expected to explain total costs including the eutrophication indicators (e.g. the concentration of ammonia present in the water). Data collected for the variables were used to estimate a relationship between the dependent variable and the explanatory variables. The regression relationship describes the effect of a change in the explanatory variable on the total cost variable and can be used to ascertain the impact of eutrophication on the particular economic sector.

Agricultural cost model

Agricultural cost data obtained from Grain South Africa (www.grainsa.co.za) were grouped into capital costs (machinery, financing, etc.) and intermediate costs (fuel, labour, etc.). A fixed effects model – a form of regression analysis that takes into account the 'individuality' of cross-sectional units by allowing the intercept term to vary for each cross-sectional unit – was used to estimate the costs of eutrophication associated with the agricultural sector for each of the three Vaal system areas. The change in total agricultural costs was estimated as a function of the change in capital costs, intermediate costs and eutrophication levels (represented by phosphorus and nitrogen levels). The model results for the three dams were relatively similar and showed nitrogen (N) levels to be statistically significant in explaining total costs to the agricultural sector. However, the estimated impact of N levels on eutrophication was relatively low; a 1% rise in N levels was estimated to increase agricultural costs by 0.005%. Eutrophication plays a significant part in the model as a whole in explaining total cost, but the effect of eutrophication is relatively small.

Industry cost model

Data collected for the industry cost model did not allow for econometric analysis. It was established, through discussions with Eskom and Sasol, that water quality in general and not eutrophication *per se*, is the dominant concern for these industries as the treatment process for salt indirectly addresses the symptoms of eutrophication. The costs incurred by water supply services due to eutrophication were used as an indirect indication of the costs of eutrophication to the industrial sector.

Water treatment cost model

Two studies within the project investigated the impact of eutrophication on water treatment costs. The first study used the costs incurred by water supply services (Rand Water) due to eutrophication as an indirect indication of the costs to industry of eutrophication. The second study investigated the relationship between raw water quality and the chemical costs of producing potable water at two water treatment plants: Zuikerbosch Station #2 (owned by Rand Water) in the Upper Vaal Water Management Area and Balkfontein (owned by Sedibeng Water) in the Middle Vaal Water Management Area.

Using an ordinary least squares regression model and data from Rand Water, the relationship between eutrophication (nitrogen and phosphorus levels) and water treatment costs was estimated. From the model results, both nitrogen (N) and phosphorus (P) levels were statistically significant in explaining total chemical costs: a 1% increase in N levels leads to a 0.2% increase in water treatment costs, similarly a 1% increase in P results in a 0.2% increase in water treatment costs for Rand Water. Based on Rand Water operating costs for 2008, a 10% rise in nitrogen and phosphorus levels would lead to a 3.2c/kL and 5.0c/kL increase, respectively, in operating costs.

To estimate the relationship between raw water quality and the chemical costs of producing potable water, a partial adjustment regression model was estimated for the Zuikerbosch and Balkfontein water treatment plants using time series data on raw water quality and chemical dosages used to treat raw water. Real chemical water treatment costs were specified as a function of real chemical water treatment costs in the previous time period and of raw water quality variables – identified through principal component analysis – in the current period.

For the Balkfontein water treatment plant, calcium and turbidity are the main drivers of chemical costs of water treatment: a 1% decrease in calcium levels would increase treatment costs by more than 3%, while an increase of 1% in raw water turbidity at Balkfontein could raise chemical water treatment costs by approximately 2%. A 1% increase in chlorophyll 'a' levels at Balkfontein would increase water treatment costs by almost 0.4%, keeping all other variables constant. These percentage changes equate to R250 000 per annum with a 1% increase in raw water turbidity and R361 000 per annum for a 1% decrease in calcium levels (at average 2004-2006 costs), provided Balkfontein treats water at its full capacity (360 000 kL/day), keeping all other variables constant.

At Zuikerbosch, water treatment costs can be predicted from four water quality variables; nitrate, total alkalinity, electrical conductivity and sulphate, and the previous period (week) cost. An increase of 1% in nitrate is predicted to increase real water chemical treatment costs by approximately 0.3% (R208 000 per annum) and a decrease of 1% in total alkalinity loading in raw water would increase real water chemical treatment cost by 0.2% (R156 000 per annum) (at average 2004-2006 costs), keeping all other variables constant and provided Zuikerbosch operates at full capacity (1 998 000 kL/day).

In an investigation by Quayle et al. (2010), this water treatment cost model was used to predict the chemical treatment costs for raw water abstracted from the Hartbeespoort, Roodeplaat and Klipvoor Dams with and without the application of zero-phosphate detergents. From the results, the application of zero-phosphate detergents could lead to a combined total predicted saving of R616 134 per year for treating water from the Hartbeespoort, Roodeplaat and Klipvoor Dams.

Property price model

A seemingly unrelated regression (SUR) model – allowing for the effect of eutrophication on the unit property price for each geographical area to be included in the one model – was used to estimate the relationship between property prices and eutrophication using secondary property price data, obtained from the Knowledge Factory. Eutrophication was proxied using levels of chlorophyll ‘a’, ammonia and nitrates and nitrites present in the water obtained from the Department of Water Affairs and Forestry (now the Department of Water Affairs).

Ammonia was estimated to be statistically significant in explaining changes to unit property prices for all three areas: a 1% increase in the level of ammonia present in the dam will subsequently reduce house prices by approximately 1% in the Bloemhof area, 0.3% in the Vaal area and 0.5% in the Grootdraai area. Both the chlorophyll and the nitrite/nitrate variable carried positive signs in several of the equations, suggesting an increase in property price with an increase in eutrophication. This outcome was unexpected, but most likely a result of averaging the eutrophication data and of the many factors (other than eutrophication) that affect property prices.

Recreation value model

A survey was conducted at each of the three dams using conjoint analysis to estimate the willingness to pay (WTP), by users of the dam, for an improvement in water quality. Conjoint analysis is a technique used to determine preferences of individuals across various characteristics of a multi-attribute choice. A total of 90 respondents returned their questionnaires: 20 respondents for the Grootdraai Dam, 30 for the Bloemhof Dam and 40 for the Vaal Dam.

Due to the relatively small sample size, it was necessary to allow for an upper and lower bound, or confidence level of the WTP values. Allowing for a 30% confidence level and rounding to the nearest rand, the following upper and lower bounds for each of the study areas were estimated:

- R25 < Bloemhof < R47
- R37 < Vaal < R69
- R46 < Grootdraai < R86, (Mostert, 2009).

These estimates can be interpreted as the average value or range of values a visitor to the respective dam would be willing to pay to improve the water quality of the dam, per night's stay.

Integrated model

A seemingly unrelated regression (SUR) model was estimated for each of the dams to simultaneously investigate the relationship between eutrophication and the various sectors (property, agriculture, water treatment and recreation) affected by eutrophication. The SUR econometric technique essentially allows for the interaction of the different economic costs associated with eutrophication in each of the three dams, within a single model. A SUR model was estimated for each of the dams for the period 1996 to 2006 and then again for the period 1996 to 2008.

The second, or expanded model, was then applied to estimate the Rand equivalent of the costs associated with eutrophication for each dam. Essentially this calculation is the actual cost of the average level of eutrophication in a given dam. The calculations of the Rand costs are limited by the level of significance of the expanded models; only costs associated with statistically significant variables are reported.

The average costs to agriculture per year, as a result of phosphorus present in the water, were estimated at R247.30/ha for the Bloemhof Dam, R454.58/ha for the Grootdraai Dam and R144.78/ha for the Vaal Dam. The property sector was most affected by chlorophyll ‘a’ levels: the average cost to the property sector was estimated as R854.10/m^2 in the Bloemhof Dam area and R3339.54/m^2 in the Vaal Dam area. The average costs to the water treatment sector per dam, as a result of ammonia levels, were estimated as R0.18/kL for the Bloemhof Dam, R0.0015/kL for the Grootdraai Dam and R0.00109/kL for the Vaal Dam.

A second application of the expanded model was undertaken to determine the costs of exceeding the Department of Water Affairs’ Resource Water Quality Objectives (RWQOs), which indicate the limit of the assimilative capacity of the dams that are within the Vaal River system (Bloemhof, Grootdraai and Vaal Dams included). The level at which phosphates and chlorophyll exceed the RWQOs represents the costs of eutrophication at large.

The results reveal that both phosphate and chlorophyll levels regularly exceed the RWQOs. The costs depend on the extent to which the RWQOs are exceeded. Here it is important to emphasise the limitation mentioned previously, namely that annual average eutrophication observations have been used. The costs of exceeding the RWQOs are, therefore, also linked to only those years where the average water quality exceeds the RWQO for an entire year. Costs, however, are linked to the variability and seasonality of rapid and short-term increases (spikes) in concentration levels of the various compounds associated with eutrophication. Irrespective of these shortcomings, it was possible to determine functional relationships between eutrophication and economic costs, especially linked to years in which the average eutrophication level exceeded the stated RWQO. The costs varied from very small to as much as R2 900/ha/year for agriculture, R1.44/kL for water treatment and R18 800/m^2 with respect to residential property prices.

However, it is difficult to comprehend the full impact of eutrophication on these sectors from the models developed in this study as the daily eutrophication data were converted to yearly averages effectively smoothing out the variability and peaks within the eutrophication data. As a result of averaging the eutrophication data, the coefficients for the eutrophication variables were likely to have been underestimated and the model results should be interpreted in the context of this limitation.

Conclusions and recommendations

1. The relationship between water quality and economic indicators is complex. Changes in water quality depend on a system of interdependent climatological and physical factors and the feedback of the culmination of these factors to impacts (and hence changes) in the economy is a slow and complicated process.

2. The estimated relationship between **agricultural costs** and eutrophication was relatively weak; however, it was shown that eutrophication did contribute to total agricultural costs. Furthermore, it is likely that this relationship was actually underestimated as a result of data and model limitations.

3. From the **water supply services treatment cost model**, the estimated relationship between water treatment costs and eutrophication (nitrogen and phosphorus levels) was statistically significant and showed that a 10% increase in nitrogen present in the water would result in an increase in water treatment costs of 3.2c/kL (1.5%). A similar increase in phosphorus present in the water would lead to a 5.0c/kL (2.3%) increase in water treatment costs.

4. The investigation into the **relationship between raw water quality and the chemical costs of producing potable** water identified total hardness, calcium content and turbidity as the main drivers of chemical costs of water treatment at Balkfontein. An increase of 1% in raw water turbidity could raise chemical water treatment costs by R250 000 per annum (at average 2004-2006 costs) and a decrease in the calcium content of raw water by 1% could add R361 000 to water treatment costs per annum (at average 2004-2006 costs), keeping all other variables constant and provided that Balkfontein treats water at its full capacity (i.e. 360 000 kL per day).

   For the Zuikerbosch water treatment plant, the study predicted that an increase in raw water nitrate in the Upper Vaal Water Management Area of 1% would increase water treatment costs by R208 000 per annum (at average 2004-2006 costs) and a 1% decrease in alkalinity would increase costs by R156 000 per annum (at average 2004-2006 costs), keeping all other variables constant and provided that Zuikerbosch treats water at a daily capacity of 1 998 000 kL per day.

5. The **property price** models showed that increased levels of ammonia present in the water would lead to a reduction in the price of property for all three study areas. However, the estimated coefficients for the chlorophyll ‘a’ and nitrite/nitrate variables were generally not statistically significant.

6. The costs to **recreation** associated with eutrophication were described using estimated willingness to pay (WTP) values for improved water quality by visitors to the three dams. Visitors to the Bloemhof Dam would be willing to pay between R25 and R47 per person per night’s stay (pppn), for water quality improvements, visitors to the Vaal Dam would be willing to pay between R37 and R69 pppn and the WTP values for Grootdraai Dam were between R46 and R86 pppn. Furthermore, many of the respondents to the Vaal (68%) and Grootdraai (45%) Dam surveys indicated concern for the water quality of these two dams.

7. The **ecosystem costs** of eutrophication were not analysed within this study (due to capacity constraints) and still require investigation. Similarly, costs of eutrophication associated with eutrophication control and monitoring (sample collection and laboratory analyses) costs as well as costs linked to eutrophication and water quality policy development remain unclear and require exploration.

8. In terms of the **integrated model**, the relationship between eutrophication and the various sectors (property, agriculture and water treatment) affected by eutrophication was difficult to establish. However, it was possible to determine functional relationships between eutrophication and economic costs, especially linked to years in which the average eutrophication level exceeded the stated Resource Water Quality Objectives (RWQO). The estimated costs vary from relatively low to as much as R2900/ha/year for agriculture, R1.44/kL for water treatment and R18 800/m^2 with respect to residential property prices.

9. The various analyses within this study show that eutrophication has an economic impact on the sectors of agriculture, property, recreation and water treatment and in a number of instances, these impacts are significant. In addition, several of the analyses underestimated the relationship between eutrophication and the particular economic sector. As a result it can be expected that the costs of eutrophication presented in this

study are even higher in reality. Furthermore, ecosystem costs of eutrophication and costs associated with the control and monitoring of eutrophication as well as costs linked to eutrophication policy and strategy development are yet to be determined.

10. The application of both the integrated model (in calculating the costs of exceeding the RWQOs) and the model of the chemical costs of producing potable water (in predicting the chemical treatment costs for raw water abstracted from the Hartbeespoort, Roodeplaat and Klipvoor Dams) demonstrate that the research undertaken in this study can be practically applied and used to inform policy and strategy development.

Even though the models specified in this study were limited in capturing the full costs of eutrophication, the results clearly show that eutrophication does impact negatively on the economic sectors considered and that these impacts are likely to be significantly higher if the full costs of eutrophication could be quantified. A more detailed discussion (Chapter 5) of several incidents of severe eutrophication, both locally and internationally, clearly illustrates the potentially significant costs of eutrophication to the various economic sectors.

It is recommended that, in addition to this study, two different kinds of analysis be undertaken in future. The one is the development of a systems-dynamic model in which both physical and economic variables are present and the linkages shown. The other is to develop a monitoring programme whereby eutrophication outbreaks are monitored. Eutrophication outbreaks are event driven, following spikes in nutrient loads. Economic data, however, by-and-large, deal with longer term variables, such as monthly on an annual basis. To improve the mapping of the impact of eutrophication it is important to monitor outbreaks properly, seek to identify its cause and effects per event and to develop an “eutrophication memory” that can be used in the medium to longer term. Such an “eutrophication memory” is likely to provide invaluable information that would improve the understanding of eutrophication considerably.

In future studies of a similar nature, effort must be made to resolve the data and model limitations outlined in Section 1.4. A possible method of overcoming these data issues is to apply ordinary spherical Kriging to the data as a spatial interpolation method (See Ara et al., 2006 for an application of this method), so as not to exclude the variability and peaks within eutrophication data from the analysis. Another option would be to model the relationship between economic sector costs as a result of severe eutrophication rather than average eutrophication levels – as it is often the worst case water quality that will drive the relationship (Dickens pers comm. 2011) – to understand the potential economic impacts of permitting eutrophication levels to rise.

Furthermore, studies of the ecosystem costs of eutrophication and the costs associated with the control and monitoring of eutrophication as well as costs linked to eutrophication policy and strategy development should be undertaken.

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LIST OF ACRONYMS AND ABBREVIATIONS

Acronym	Explanation
Chl-a	Chlorophyll 'a'
CVM	Contingent valuation method
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
HP	Hedonic pricing
N	Nitrogen
NH_4	Ammonia
NO_2NO_3	Nitrite/nitrate
P	Phosphorus
PO_4	Phosphate
RWQO	Resource water quality objectives
SUR	Seemingly unrelated regression
TCM	Travel cost method
WRC	Water Research Commission
WTP	Willingness to pay

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GLOSSARY

Conjoint analysis: a technique used to determine individual's preferences across various characteristics of a multi-attribute choice.

Contingent valuation: a survey-based economic technique for the valuation of non-market resources, based on the idea of knowing people's willingness-to-pay for certain characteristics of their environment by asking them directly.

Fixed effects regression: a form of regression analysis used with panel data that takes into account the 'individuality' of cross-sectional units by allowing the intercept term to vary for each cross-sectional unit. The intercept term remains fixed over time.

Hedonic pricing: is a revealed preference method of estimating demand or value by decomposing the price/value of an item into separate components. Estimating hedonic prices makes it possible to identify the extent to which specific components affect the price/value.

Ordinary least squares regression: a method of estimating the unknown parameters in a linear regression model. The method minimises the sum of squared vertical distances between the observed responses in the dataset and the responses predicted by the linear approximation.

Panel data: combines features of both time series and cross-section data.

Primary data: data collected by the researcher/user for the purpose of the study.

Regression analysis: the study of the dependence of one variable, the *dependent variable*, on one or more other variables, the *explanatory variables.* It is a method of modelling the relationships among three or more variables to predict the value of one variable given the values of the others.

Secondary data: data collected by someone other than the current researcher/user.

Seemingly unrelated regression (SUR): is a regression model that includes several individual or underlying relationships, linked by the fact that their disturbances are correlated. The SUR model allows for the interaction of the different economic costs associated with eutrophication in each of the three dams, within a single model.

Serial correlation: the correlation between the error terms from different time periods. The error terms are not independent of one another. Serial correlation affects the efficiency of ordinary least squares regression estimators.

1. INTRODUCTION

1.1 Project background

By 2008, approximately 35% of the total water storage available in South African dams had deteriorated in water quality due to excessive nutrient loading (Van Vuuren, 2008:14). Eutrophication, which is the enrichment of the water environment with plant nutrients, causes excessive growth of phytoplankton (free floating algae) and rooted macrophytes. The presence of large numbers of phytoplankton in water bodies causes problems such as increased water treatment costs, taste and odour problems in treated drinking water, potential health risks, interference with recreation and a reduction of the amenity values and conservation status of water resources. As such it has negative effects and cost implications for economic sectors as diverse as providers of potable water (treatment cost, health risks and consumer acceptance), industry (treatment costs), agriculture (treatment cost and toxicity to livestock), real estate (loss of property values), recreation (aesthetics, interference with boating, swimming and fishing) and for ecosystems (changes to biodiversity). Eutrophication thus diminishes the quality of water resources for many users and costly treatment is often required to overcome its negative effects. The costs associated with eutrophication need to be estimated for different levels of eutrophication, in order to consider and justify often expensive preventative measures.

Eutrophication may result in algal blooms that can impact significantly on water quality producing taste, odour and toxicity problems in treated potable water. However, quantifying these impacts is not a simple task, since their severity is dependent on a number of variables such as:

- Algal species;
- The algal cell concentrations;
- Water quality;
- The abstraction conditions (variable extraction points);
- State of cells entering the water treatment works (intact or lysed);
- Treatment train or processes at the waterworks;
- The extent and duration of the bloom;
- The size of the water body in which the bloom occurs; and

- Local conditions, such as weather, climate, topography.

Additionally, the algal cell concentration at which taste and odour compounds and algal toxins become problematic may vary, even within the same water body. Cell concentrations of a particular species that might constitute a taste, odour or toxin problem on one occasion, will not necessarily result in any problems on another occasion. This is despite there being no apparent significant differences in any of the variables known to affect the growth of these algae.

The quantification of the costs associated with eutrophication will assist researchers and policy makers in identifying appropriate policies towards addressing eutrophication problems in South African river systems and will inform river catchment management decisions to remedy eutrophication problems.

1.2 Project objectives

The objectives of the project were as follows.
Generally:

- To develop a generic model to assess the costs associated with eutrophication in South Africa and to apply it to the Vaal River system.

Specifically,

- To identify and modify a first order model to link the level of eutrophication to its primary drivers.
- To develop a generic model with which the direct and indirect cost of eutrophication to different economic sectors and ecosystems can be assessed for South African conditions.
- To demonstrate the applicability of the model by applying it to the Vaal River as a case study.
- To apply the model to calculate the direct and indirect costs associated with present and realistic future nutrient level/eutrophication scenarios.

1.3 Project approach

The approach of the project was to develop a generic model to assess the economic costs associated with eutrophication (1) through a comprehensive literature review and (2) through a desktop analysis of primarily existing data to develop the model and (3) to apply the model to data from the Vaal River system.

The project was initially set out as part of a PhD study and the bulk of the research efforts and resource economics modelling were to have been part of the PhD study. However, due to changing circumstances the project had to be re-organised and conducted in three phases. The first phase comprised two MSc studies that investigated the cost of eutrophication to the agricultural and water treatment (potable water) sectors (de Villiers, 2009) and the residential and recreation sectors (Mostert, 2009) respectively. The second phase was a third MSc study on the development of an integrated model using the results of the studies obtained in the first phase (Sibande, 2010). The third phase entailed the development of a synthesis report of the three MSc studies. Secondary data – data not collected by the researcher, but obtained from alternative sources – were primarily used in the MSc studies. A survey was conducted by Mostert (2009) that investigated the effects of eutrophication on the recreation sector which produced primary data for that section of the project.

This report is a summary and discussion of the results of these three phases as well as additional relevant literature and studies within the scope of this project. The MSc theses are available on the CD accompanying this report.

It was initially intended to investigate the ecosystem costs of eutrophication within this study and this objective fell under the original PhD project. However, as a result of the restructuring of the study, these costs were not analysed due to a lack in capacity to address these. Similarly, costs of eutrophication associated with eutrophication control and monitoring (laboratory) costs as well as costs linked to eutrophication and water quality policy and strategy development remain unclear and require exploration. The costs to industry were estimated indirectly using the costs incurred by water supply services (water treatment costs) due to eutrophication.

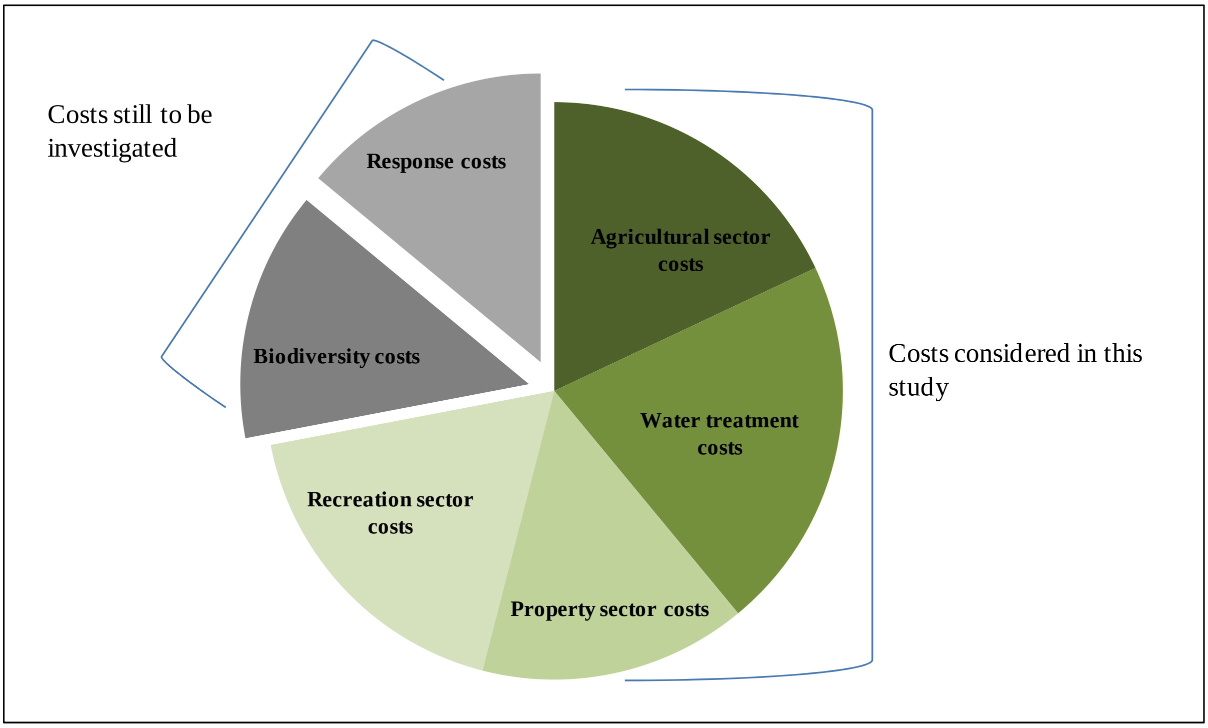

Figure 1.1: A representation of the costs of eutrophication modelled in this study and those costs still requiring investigation.

Owing to the separation of the original project into several theses, the project shifted from an initially integrated evaluation of the total costs of eutrophication to a more statistical approach of estimating the costs associated with eutrophication. As the development of the models progressed, it become clear that there are several layers of costs associated with eutrophication and that the statistical models were unable to capture the complete economic costs of eutrophication. An additional discussion was added to the study to further develop the concept of eutrophication costs and to identify how the costs estimated by the statistical models feature in terms of the total economic impact of eutrophication.

1.4 Project limitations

Eutrophication is an umbrella term for the combined adverse effects of several water quality indicators such as phosphate and nitrate concentrations and their impact on the quality of water in a water body, commonly reflected as elevated chlorophyll 'a' concentrations. Typically, eutrophication data present high levels of variability within a specific time period (see for example Figure 1.2). The high levels of variability and significant level changes over time within the eutrophication data create challenges for the econometric analysis of the economic costs associated with eutrophication. This is because the economic data collected is typically annual and, by definition, very coarse. Eutrophication data, on the other hand, may be daily measurements at specific times and at specific geographical points.

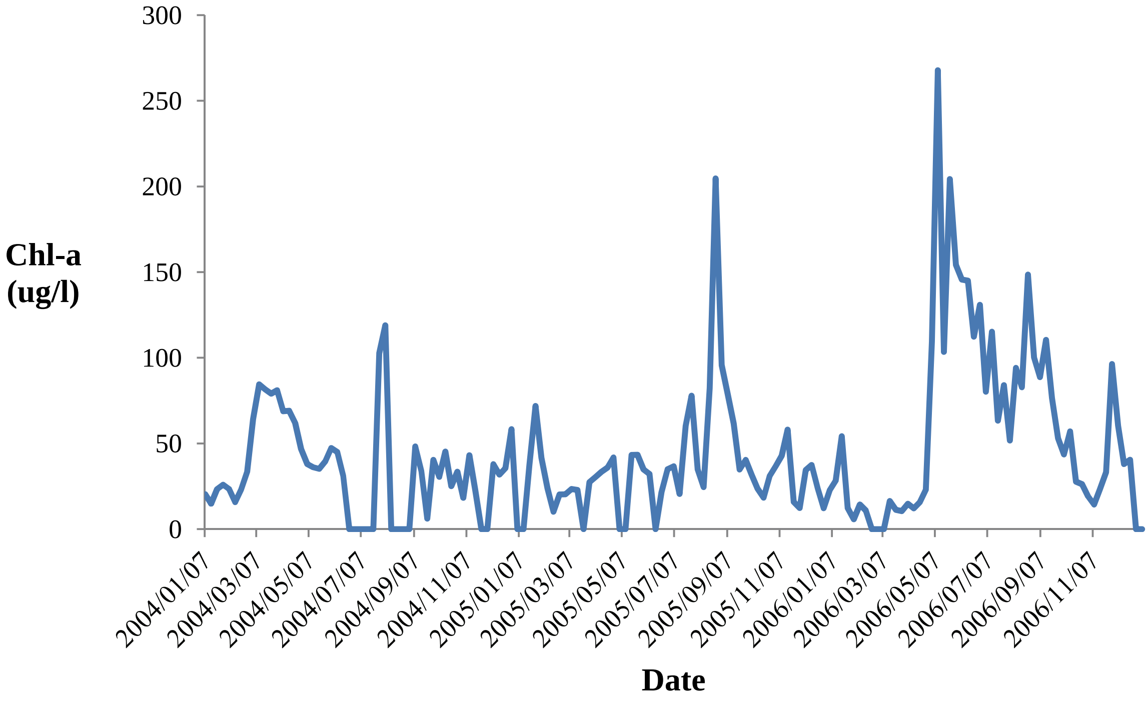

Figure 1.2: Graph of weekly chlorophyll ‘a’ levels at Balkfontein, 2004-2007.

Source: Sedibeng Water (www.sedibengwater.co.za/)

To conduct econometric analyses, the daily observations of concentration levels (eutrophication data) must be transformed to annual averages. This means that the variability and peaks within the eutrophication data are effectively smoothed out. However, the ecological and economic consequences of eutrophication are most often experienced as a result of changes in the level of eutrophication and the variability thereof at the time the variance occurs. For example, potable water treatment plants are typically able to cater for and treat the results of eutrophication occurring in the abstracted water. However, during times of extreme eutrophication the algae present may produce high concentrations of taste and odour compounds. This has a significant impact on water treatment costs through having to employ additional sophisticated and expensive treatment measures such as dosing the water with powered activated carbon. Figure 1.3 is a graphical representation the changes in costs associated with eutrophication.

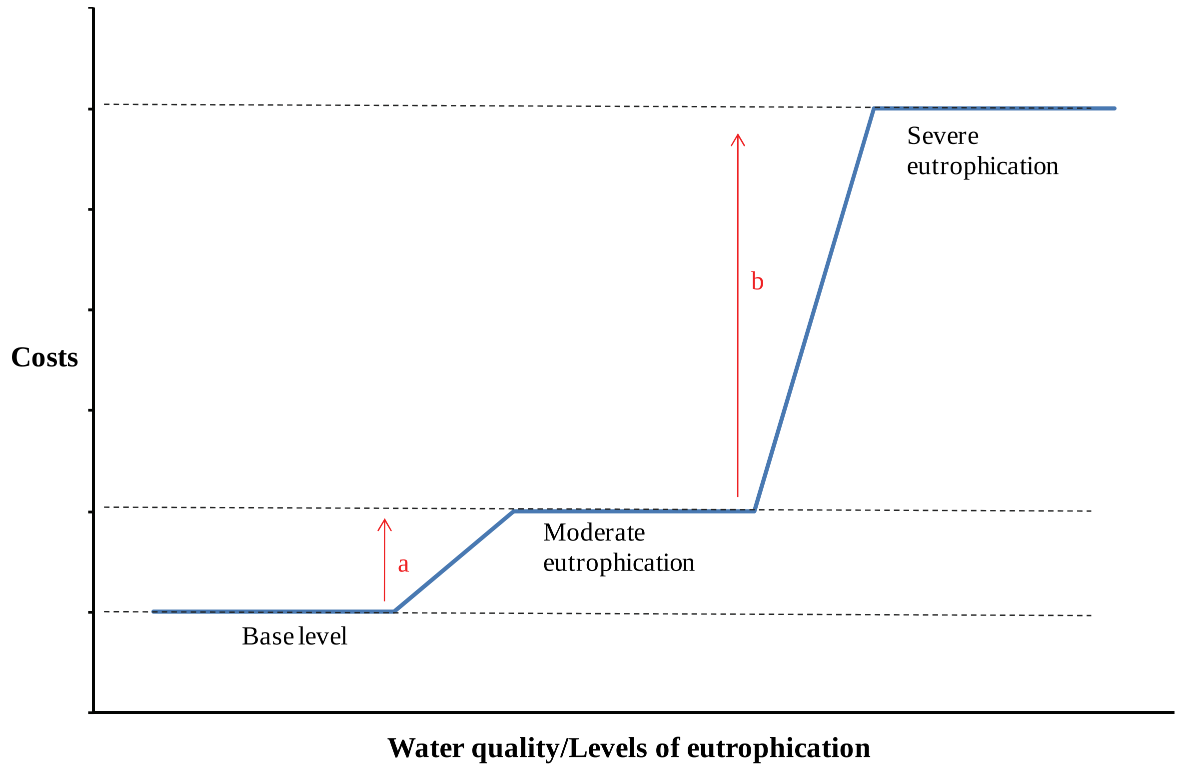

Figure 1.3: Illustration of the expected relationship between eutrophication and its associated costs.

Due to the averaging of the eutrophication data, the costs associated with eutrophication estimated within this project are effectively the costs of eutrophication within "bands" of eutrophication levels (base, moderate and extreme eutrophication). The costs associated with changes, or peaks, in eutrophication (the "jumps" a and b in Figure 1.3) are effectively missed out from the econometric analyses conducted within this project. Note the significant 'jump' (b) in costs associated with levels of extreme eutrophication. As a result of averaging the eutrophication data, the relationships estimated between eutrophication and the various economic sectors are likely to be relatively weak and the model results (Chapter 4) should be interpreted in the context of this limitation. Even though the models specified in this study were limited in capturing the full costs of eutrophication, from the results it is clear that eutrophication does negatively impact the economic sectors considered and that these impacts are likely to be significantly higher if the full costs of eutrophication could be quantified. Chapter 5 provides a discussion of the full costs of eutrophication and identifies which of these costs have been captured by the models applied within this project.

1.5 Report structure

The next chapter of the report is a review of the literature. Chapter three describes the methodology and data used in the project and draws on the MSc dissertations by Gebremedhin (2009), de Villiers (2009), Mostert (2009) and Sibande (2010). The fourth chapter presents the results from the various studies along with the application of the integrated model to the Vaal River system. Chapter 5 is a discussion of the cost associated with eutrophication. The report closes with conclusions and several recommendations for future research.

2. LITERATURE REVIEW

2.1 Introduction

Eutrophication is defined as the enrichment of water by nutrients, stimulating an array of symptomatic changes including increased production of algae and/or higher plants, which can adversely affect the diversity of the biological system, the quality of the water and the uses to which the water may be put (Pretty et al., 2002). It is primarily caused by high loads of phosphorus and nitrogen in water bodies, which may be the result of natural processes of nutrient cycles and biogeochemical processes in the soil and atmosphere (natural eutrophication), or as a consequence of water pollution from domestic, industrial and agricultural activities augmenting naturally occurring nutrient levels (cultural eutrophication) (UNEP, 2005).

Amongst others, symptoms of eutrophication include an increase in water turbidity, a changed composition of the algal flora, an increased frequency of anoxic situations and, possibly, more algal blooms (Söderqvist and Scharin, 2000). The impacts of eutrophication on the natural biodiversity and ecology of river systems include increased fish and invertebrate mortality, increased mortality and morbidity of livestock, increased occurrence of human health problems, reduced amenity and recreation value of waterside property (particularly if the water becomes turbid and there is emission of unpleasant odours from algal blooms) and increased costs of treating raw drinking water. Thus, eutrophication has negative consequences for the conservation status of water resources, industry, agriculture, real estate values, recreation and tourism, the provision of potable water and public health costs (Walmsley, 2000; UNEP, 2005).

Eutrophication reduces water quality and costly treatment is often required to overcome its negative effects. This creates problems for economic development and sustainable economic growth (Dennison and Lyne, 1997). Growth in the demand for clean water due to growth in real per capita incomes, population growth and urbanisation has exacerbated this problem over time (Markandya and Richardson, 1992; UNEP, 2005). Consequently, eutrophication is regarded as a serious environmental problem and can pose a major challenge to sustainable economic development in South Africa (SA) and globally.

Several studies in the United States (US) and Europe have valued economic costs and benefits of reduced eutrophication (D'Arge and Shogren, 1989; Dixon et al., 1994; Zylicz et al., 1995; Michael et al., 1996; Sandstrom, 1996; Legget and Bockstael, 2000; Poor et al., 2001; Gibbs, 2002; Elofsson, 2002; Batalhone et al., 2002; Soutukorva, 2005). Most of these studies focused on finding cost effective ways of nutrient load reduction. For example, Elofsson (2002) calculated cost effective solutions to nutrient load reductions to the Baltic Sea, while taking into account the stochastic relationship between abatement measures and the corresponding impact on nutrient loads. There are, however, only a few published studies on the economic costs of water eutrophication in SA, e.g. Dennison and Lyne (1997) and Van Zyl and Leiman (2002). Dennison and Lyne (1997) investigated water treatment costs in relation to water quality in the Umgeni River catchment in KwaZulu-Natal (KZN).

The Vaal River has been described as the backbone of South Africa's economy since it provides water services to the economic hub of South Africa, the Gauteng Province (Tempelhoff, 2006:433). The Vaal River system encompasses a number of bulk water storages (reservoirs, dams), the largest of these being (in downstream order) Grootdraai, Vaal and Bloemhof. The Vaal River supports the production of 25% of the Gross Domestic Product of the South African economy and over 12 million people depend directly on it for water (Tshwane University of Technology, 2009:37). However, it has become apparent that the Vaal River is increasingly under threat of pollution and significant intervention is needed to reverse this trend (Sibande, 2010).

The principal use of water from the Vaal River system is for domestic and industrial purposes (including mining and for cooling electricity power stations). The Vaal River system is also used extensively for recreational and amenity purposes, such as swimming, fishing and sailing (Bruwer et al., 1985). The fishing industry alone generates R1.3 billion a year along the Vaal River (Farmers Weekly, 2006). Eutrophication varies geographically within the Vaal River system, ranging from infrequently severe to persistently severe (Walmsley, 2005, Appendix A). Although Mirrilees et al. (2003) estimated the economic value of water for the Vaal River system, as far as the authors are aware, no research on the costs of eutrophication in the Vaal River system has been published to date.

2.2 Primary drivers of eutrophication

In 1982, the Organisation for Economic Cooperation and Development (OECD) defined eutrophication as "… the nutrient enrichment of waters which results in the stimulation of an array of symptomatic changes, amongst which increased production of algae and aquatic macrophytes, deterioration of water quality and other symptomatic changes are found to be undesirable and interfere with water uses". Other definitions, such as the one used by the US Geological Survey (USGS, 2011), "Eutrophication is a process whereby water bodies, such as lakes, estuaries, or slow-moving streams receive excess nutrients that stimulate excessive plant growth (algae, periphyton attached algae, and nuisance plants weeds)", mirror the original OECD definition.

These definitions encompasses both a *process* – the collective actions leading to the enrichment of water bodies with nutrients – and a *response* – the environmental change induced in the receiving water body by the increased availability of nutrients.

Eutrophication is an ecological term that was originally used to describe the natural process by which a water body becomes enriched with plant nutrients. During this process the water body accumulates organic matter (both living and decaying matter) and progressively changes from that of a deep water body (such as a lake) to that of a wetland and, ultimately, to that of a terrestrial system. The term was originally associated with the process of natural ageing of lakes. Under natural conditions this process takes place over tens of thousands of years. However, human influences have greatly accelerated the rate of enrichment and today a distinction is made between natural eutrophication and cultural eutrophication.

- Natural eutrophication is a natural process that is dependent on the geology and natural features of the catchment. It is not reversible and continues at a slow rate.
- Cultural eutrophication refers to the human-induced process that is related to anthropogenic activities in the catchment. It is associated with human social and economic activities, and accelerates the rate of ageing of a water body. Cultural eutrophication is reversible. In this document the term "eutrophication" is used to refer to cultural eutrophication rather than natural eutrophication.

2.2.1 Nutrient enrichment and eutrophication

Nutrients are mostly inorganic elements that are assimilated by plants and, in conjunction with the process of photosynthesis, are utilised to produce and accumulate organic material in aquatic ecosystems. Algae and other aquatic plants require about 20 different elements. However, the rate and extent of aquatic plant growth is dependent on the concentration and ratios of nutrients present in the water. Plant growth is generally limited by the concentration of that nutrient that is present in the least quantities, relative to the growth needs of the plant. This is known as the limiting nutrient concept and is the basis of eutrophication management policies. Because of nutrient supply and demand in nature, it has been observed that phosphorus (P) and nitrogen (N) are the most frequent limiting nutrients in freshwater systems; P being the primary limiting nutrient and N the secondary. Increases in the levels of either of these two nutrients in a water body will raise the risk (and frequency) of experiencing eutrophication problems. Management of N and P inputs to the aquatic environment is, therefore, the key to the eutrophication problem.

Nitrogen occurs in surface waters in several forms (e.g. ammonia, nitrite, nitrate, urea, and nitrogen gas). All freshwater algae are able to assimilate the first four forms, but nitrogen gas can only be utilised by certain species of blue-green algae (cyanobacteria such as *Anabaena* species).

Phosphorus occurs as orthophosphate, polyphosphates and organic phosphates. Orthophosphate is generally considered to be the most immediately available form of P. However, mineralisation, adsorption onto suspended material or sediment, desorption under aerobic conditions, luxury uptake by plants and precipitation with calcium or iron, are all processes that play important roles in influencing the concentration of available P in fresh waters. Figure 2.1 summarises the main causes of eutrophication.

Biological growth and decay processes in aquatic ecosystems are important in determining the levels of available nutrients. Plants accumulate nutrients whilst decomposition of dead plant material by bacteria mineralises them. These processes occur at different rates. Real time observations of ambient concentrations can, therefore, lead to an incorrect understanding of the level of nutrients. A eutrophic ecosystem is one where the *potential* concentration of nutrients is excessive even when instantaneous concentrations might be low.

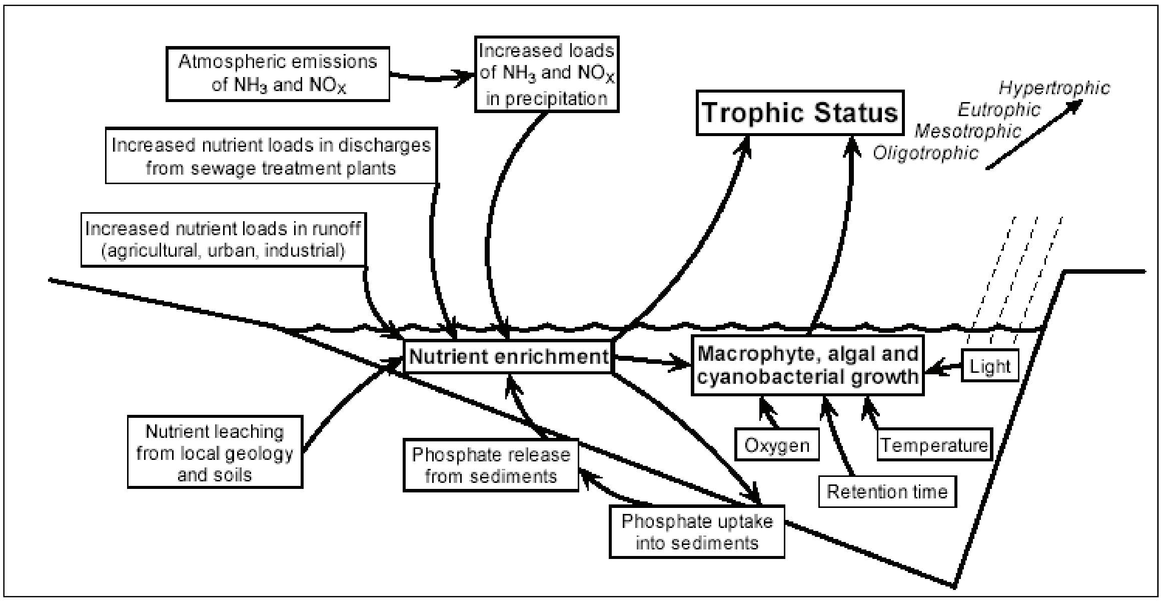

Figure 2.1: Diagram of the main causes of eutrophication.

Source: DWAF, 2002

2.2.2 Sources of nutrients and pathways

The eutrophication process can best be understood by examining nutrients, their sources and their pathways to the aquatic environment. Knowledge of both the N and P cycles is also important as these cycles influence the availability of these two nutrients. The main driving force to the eutrophication problem is human population growth and associated economic activities. Human activities make use of numerous products and resources (containing bound N and P), ultimately converting them into elemental compounds (available N and P) that are released into the aquatic environment through various pathways.

There are numerous products that act as primary sources of N and P. Examples of these include:

- Powder detergents which have a high P content and are used for domestic and industrial cleaning. These enter the aquatic environment via wastewater treatment discharges.
- Foodstuffs (e.g. meat, cereals, milk, etc.) that are part of the everyday consumer economy all contain N and P which is ultimately discharged into the environment via sanitation and sewage treatment systems.
- Fertilisers that contain high levels of N and P (and other plant nutrients) are applied to agricultural land to stimulate crop productivity. Most fertiliser applications are well in excess of actual requirements and are an important source of nutrients. These enter

the aquatic environment via washoff from fertilised agricultural lands and in irrigation return flows.

- Fossil and wood fuels such as coal, oil and wood that are used for transport and power generation contribute to atmospheric discharges of nitrogen oxides that are then deposited on to land and water surfaces.
- Materials such as fishmeal, hay and grass (and supplements) that are supplied to animals in feedlots all have a significant N and P content. The animals consume the material and their wastes, which are high in ammonia and phosphate, are then discharged into the aquatic environment. Ammonia is also emitted into the atmosphere and reaches the aquatic environment through atmospheric deposition.

There are two main pathways in which nutrients are introduced to the aquatic environment and these can be classified as point or diffuse (non-point) sources (Table 2.1). A point source is usually a location of high nutrient concentration (e.g. a factory, domestic wastewater plant, an animal feedlot) where some form of effluent is discharged (via a pipe) directly into a receiving aquatic system (river or reservoir). A diffuse source is a location with multiple nutrient sources that are spread over a much wider area and nutrients enter the aquatic environment through leaching, washoff and atmospheric deposition processes (e.g. an agricultural field, atmospheric depositions, a township or an urban area). For diffuse sources the nutrients are transported over the landscape and come into contact with soils and vegetation before entering the aquatic environment.

Table 2.1: Typical characteristics of point and non-point sources of water pollution.

Point Sources	**Nonpoint Sources**
Relatively steady flow over time	Flows usually occur at random and intermittent intervals following rainfall events
Adverse impacts are most severe during periods of low flow in receiving streams or cumulative in receiving reservoirs	Adverse impacts most severe during or following storm events or cumulative in reservoirs
Pollutants enter water courses at identifiable points	Pollutants enter water courses at many, often unidentifiable points

It is generally accepted that, because of containment volumes and concentration characteristics, nutrient management and control is much more easily implemented at point sources rather than diffuse sources.

During their transport from primary sources to the receiving water body, all nutrients undergo transformations that are associated with P and N cycles that change their concentration, chemical form and availability. It is well documented that not all of the potential supply of N and P eventually ends up in the receiving aquatic environment. The longer the distance (and time) between the source (e.g. factory, field, feedlot, township) and the water body of concern, the lower will be the amount of nutrients that contribute to eutrophication problems.

2.3 Impacts of eutrophication

Eutrophication is often a consequence of water pollution from domestic, industrial and agricultural activities augmenting naturally-occurring nutrient levels. In a market economy the decisions of households and industries are based only on the expected private economic costs and benefits of their options. However, the activities of one industry or household may incur additional beneficial or adverse consequences for other industries or households known as production externalities. Polluters often do not bear the full cost of the negative externality they generate and will, therefore, continue to engage in an excessive amount of polluting from the perspective of society (Field, 1997; UNEP, 2005). Figure 2.2 is an illustration of the negative impacts of eutrophication.

The symptoms of eutrophication (algal blooms, unsightly scums, tastes and odours in drinking water, etc.) are often the aspects that are most closely related to the costs incurred in order to deal with those symptoms. There are a number of variables that are used as indictors of eutrophication. These include biological, chemical and physical indicators. Appendix B provides a brief description of a number of these indicators of eutrophication.

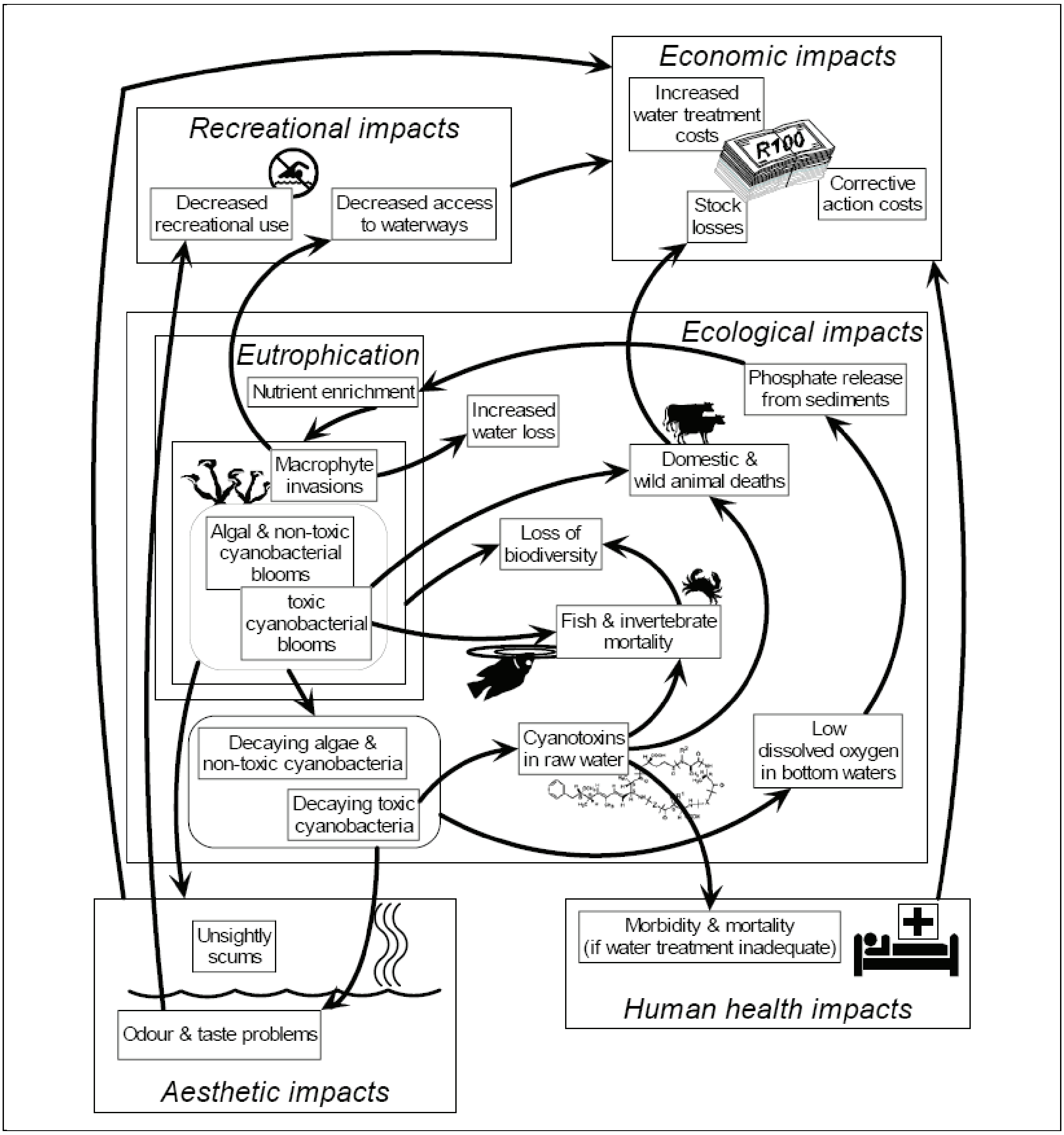

Figure 2.2: Illustration of several of the negative impacts of eutrophication.

Source: DWAF (2002)

2.3.1 Costs of eutrophication

Adopting the approach of Pretty et al. (2002), the costs of eutrophication may be broadly categorised as damage costs and policy response costs. Damage costs refer to the reduced value of the water, and may be further sub-divided into social damage costs and ecological damage costs. The policy response costs refer to the costs of addressing and responding to eutrophication and may also be divided into two types of costs, i.e. compliance control costs (arising from adverse effects of nutrient enrichment) and direct costs (incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication) (Pretty et al., 2002). Figure 2.3 is a summary of the categorisation of eutrophication costs.

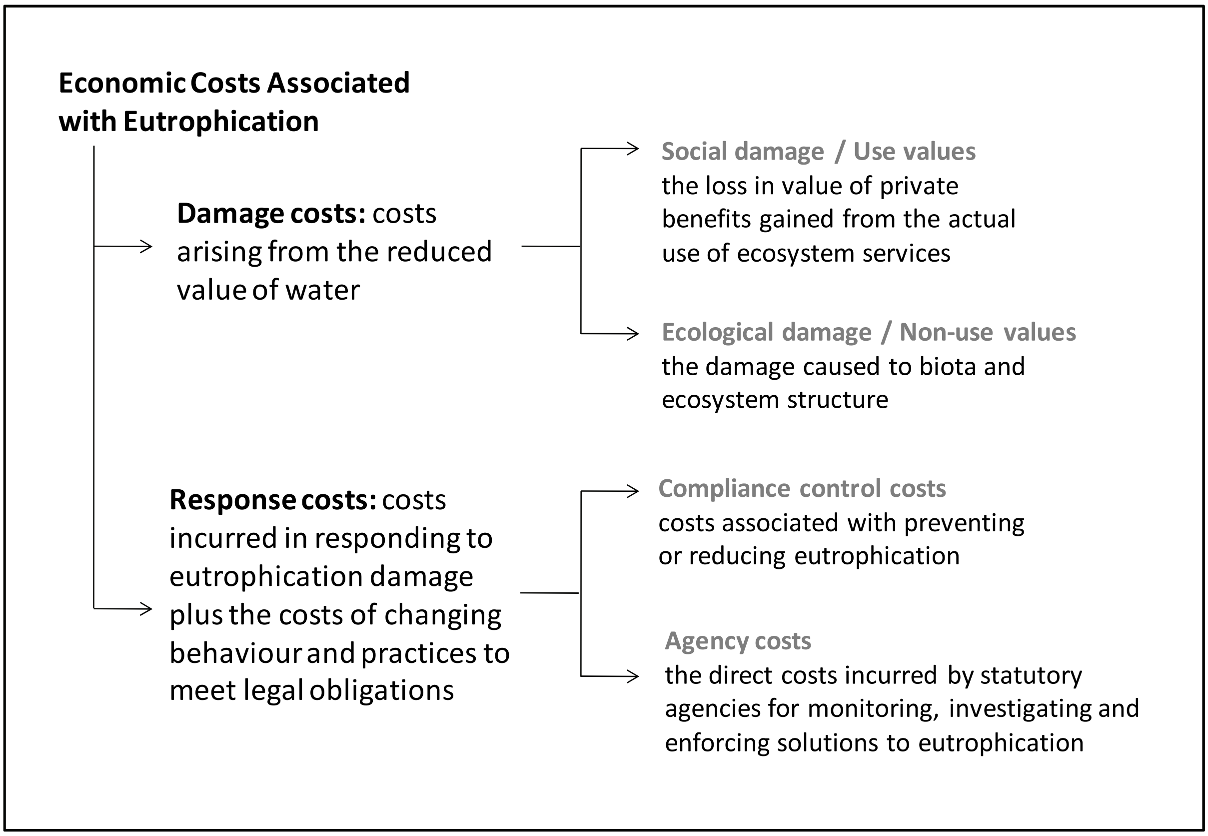

Figure 2.3: Categorisation framework of the economic costs associated with eutrophication.

Source: Adapted from Pretty et al. (2002)

2.3.1.1 Damage costs arising from eutrophication

Damage costs of eutrophication represent a loss of existing value, rather than an increase in costs, and are divided into two categories: use values (social damage costs) and non-use values (ecological damage costs). Use values are associated with private benefits gained from the actual use of ecosystem services and can include private uses such as agriculture, industry, recreation benefits, education benefits, general amenity benefits and option values (Pretty et al., 2002).

2.3.1.1.1 Social damage costs (use values)

Eutrophication can adversely affect the use value of water bodies, such as reduced amenity and recreation value of water side properties; reduced values of water bodies for commercial uses (in agriculture and industry); drinking water treatment costs; clean-up costs of waterways (dredging and aquatic weed control); and reduced recreational and amenity value of water bodies for water sports.

Reduced value of water side property

Residential properties with a waterfront tend to have higher values than equivalent properties without a waterfront (Braden and Kolstad, 1991; Michael et al., 1996; Pretty et al., 2002). Water quality can be an important determinant of this premium and a change in water quality can have a significant impact on property values if property market participants are aware of water quality and the impact of water quality on the utility derived from amenity and recreation activities on waterfront properties. Many water quality indices used by natural scientists reflect levels of pollutants that are not easily observed or that do not impair the utility derived from waterside properties. Thus, homeowners and recreationists are often unaware of water quality until high nutrient concentrations combine with other factors to cause algal blooms, which reduce water clarity and can cause large algal scums on the surface of the water (e.g. Legget and Bockstael, 2000).

Hedonic Pricing (HP) is an important approach for estimating the impact of eutrophication on the value of waterfront properties. In HP, property values are regressed on property characteristics to estimate the partial contribution of each characteristic to property value (Van Zyl and Leiman, 2002). The use of HP to capture the effect on price of improvements in environmental quality, therefore, requires observations on sufficiently varying water quality levels within the confines of a single housing market (Legget and Bockstael, 2000). For this reason, only a few studies have used HP to study the impact of water quality on the values of water front properties during the past three decades (Legget and Bockstael, 2000).

An example of the application of HP to determine the impact of eutrophication on waterside property values is the study by Poor et al. (2001) to determine the impact of water clarity on lake front properties on selected fresh water lakes in Maine, US. Poor water clarity in the Maine Lakes is primarily attributable to non-point source pollution as water clarity is reduced as the concentration of algae in the water increases. Water clarity is a relatively straightforward dimension of water quality upon which directly comparable objective and subjective measurements can be obtained. Consequently, it is commonly used as a measurement of eutrophic status by limnologists (Poor et al., 2001).

Reduced recreational and amenity value of water bodies

Many fresh and marine water bodies are used extensively for recreational and amenity purposes. Eutrophication may result in the loss of recreational and amenity value, particularly if water becomes turbid, emits unpleasant odours and is affected by algal blooms (Pretty et al., 2002). Value loss costs are incurred, not only when people are prevented by eutrophication and algal blooms from enjoying the quality of the water body, but also by those people whose livelihoods rely on visitors who would otherwise have used the clean water body. When visitors access water bodies for recreational purposes, money is spent on accommodation, food and other goods and services. Consequently, when eutrophication prevents access, this revenue is lost (Pretty et al., 2002). This results in losses for these locations and negative effects on the tourism industry. Economists have conducted a number of studies using a variety of methods to put a value on the recreational uses of water bodies. Contingent valuation, travel cost, and hedonic pricing methods are some of the most familiar approaches to measure recreational and amenity value loss from water bodies (Zylicz et al., 1995; Michael et al., 1996; Soutukorva, 2005).

Human health and livestock watering costs

Increased mortality and morbidity of humans and livestock attributable to direct use of water from a polluted water source reduces the use value of the water (Pretty et al., 2002). Diminished water quality due to eutrophication can produce a variety of adverse health effects, ranging from slight pain to acute episodes requiring hospital care and even death. The value of this cost can be estimated from (a) the observed costs incurred by households to avert these outcomes, (b) from contingent valuation method (CVM) studies of peoples' expressions of their willingness-to-pay to avoid these outcomes (e.g. Le Goffe, 1995; Söderqvist and Scharin, 2000), and (c) from industrial wage rate studies (Field, 1997).

Water treatment costs

One of the key manifestations of eutrophication is an increase in algal blooms in dams and lakes. Often, the dominant algae associated with this increase are blue-green or cyanobacteria with the ability to produce potent taste and odour compounds and even algal toxins. It is these aspects of eutrophication which have a significant impact on water treatment, drastically increasing costs, as drinking water must be treated to remove algal toxins and algal decomposition products and to remove nitrogen. Treatments and actions aimed at removing algal toxins and algal decomposition products may be highly significant costs for water supply and water treatment operators. Water treatment costs are also incurred by water supply companies to comply with drinking water standards as well as by sewerage treatment operators to meet effluent discharge compliances and regulations.

A range of methodologies have been used to estimate water treatment costs in the literature reviewed. Pretty et al. (2002) estimated drinking water treatment costs by water treatment companies using ordinary least squares (OLS) regression to estimate the operating and capital costs of removing nitrogen from water. Dennison and Lyne (1997) applied a different approach to a similar problem, i.e. they estimated the drinking water treatment cost for the DV Harris water treatment plant using the partial adjustment regression model. The approach used by Dennison and Lyne (1997) seems very admissible in that water treatment costs incurred in one time period are a function of nutrient loads in previous periods; a partial adjustment regression model is, therefore, applicable in such a study. Dennison and Lyne (1997) show that there is a steady increase in the real treatment costs of water over time. The real treatment costs had increased from an average of R0.024 /kL to about R0.028 /kL during their study period.

Clean-up costs of waterways

Clean-up costs are the costs incurred by organisations or agencies charged with the responsibility for keeping waterways clear of weeds and dredging the nutrient-rich sediments in these waterways. According to Pretty et al. (2002), organisations in the UK may incur these costs twice yearly to clear the weeds in eutrophic waters. The high cost of clean-up may be due to the intensive labour requirement for mechanical removal of the weeds and chemical treatment. If the nutrient rich mud is disposed of on arable land the benefits gained may trade off to some of the costs incurred for dredging. The cost can be measured from annual operating and capital costs of removal of weeds and dredging the nutrient rich mud. The econometric technique that might be applied to estimate these costs, is OLS multiple regression analysis.

Hyacinth is the most successful invader weed of fresh water, nutrient enriched, eutrophic water bodies and has a high rate of vegetative growth and multiplication (Gopal, 1987). It is associated with the death of fish and the proliferation of agents of several deadly diseases. Gopal (1987) indicated that where the plant produces seeds, the seeds may cause a new outbreak of water hyacinth even after a site is completely cleared of an initial infestation.

This is a typical problem in the upper Vaal River at Parys and Potchefstroom (Walmsley, 2005).

2.3.1.1.2 Ecological damage costs (non-use value)

Negative ecological effects comprise the damage caused to biota and ecosystem structure by eutrophication. Often these values do not have market prices and so costs are difficult to assess. Eutrophication of water bodies has a direct effect on the primary production of plants and indirectly changes the abundance and nature of the organisms within the water body (Pretty et al., 2002). In eutrophic waters, a variety of changes disturb the normal ecosystem. For example, the availability of light and oxygen to other organisms is reduced with large blooms of blue-green algae. This in turn may increase the turbidity and prevent submerged and rooted macrophytes from growing (Pretty et al., 2002). The value loss costs related to intrinsic value (non-use) of species and ecosystems caused by eutrophication are difficult to measure. However, some studies indicate that the cost of restoring the affected species and habitats could be used as a proxy for the eutrophication costs (Pretty et al., 2002).

2.3.1.2 Response costs

Policy response costs are defined as "the costs of addressing and responding to eutrophication" (Pretty et al. 2002). Policy response costs are divided into compliance control costs and direct costs incurred by agencies. Pretty et al. (2002:12) hold that compliance costs include sewage treatment costs (i.e. reducing nutrient concentrations in effluent discharges), costs of treatment of alga blooms, and cost of adopting new farming practices to emit fewer nutrients. Costs incurred by statutory agencies are comprised of monitoring costs for water and air (e.g. costs of monitoring, investigating and enforcing solutions to eutrophication) and the cost of developing and implementing eutrophication control policies and strategies (Pretty et al., 2002). Assessing water quality conditions and identifying impairments by nutrients and algae in water resources entails high resource (monetary) costs. This typically necessitates high costs by governmental and statutory monitoring and water management agencies (e.g. USEPA, 2000; Pretty et al., 2002). Training programs that build expertise on water issues among government staff, water users, and decision-makers are essential. The effective protection of water resources and ecosystems from pollution requires considerable upgrading of most countries' present capacities, including the minimum infrastructure (e.g. laboratories) and staff to identify and implement technical solutions and to enforce regulatory action (UNEP, 2005).

2.3.2 Benefits of eutrophication

2.3.2.1 Fertilisation effects on farm land

Eutrophication involves a rise in plant productivity as a result of an increased availability of nutrients, mainly nitrogen and phosphorus, in water. The use of nutrient enriched water for irrigation purposes may be considered as a benefit of eutrophication. Pretty et al. (2002) established that 55 000-165 000 tones of nitrogen could be obtained from the 24 288 ML of water abstracted annually for irrigation. Therefore, farmers would benefit by substituting nutrient enriched irrigation water for purchased fertiliser.

2.3.2.2 Increase in the value of fresh water fisheries

An increased volume of fish and aquatic habitat is also one of the benefits gained by eutrophication according to Pretty et al. (2002). However, eutrophication may have several complex effects on fish populations, for example it benefits certain species and disadvantages others (Hansson et al., 2000). Similarly, when eutrophication occurs, more organic matter must be broken down, with the result that more oxygen is consumed. This could result in relatively lower levels of dissolved oxygen in deep water, which may cause a decline in deep water species of fish (Hansson et al., 2000).

2.3.2.3 Improved sources of food for wild life

Eutrophication may in some instances lead to increases in biodiversity. For example, birds are particularly attracted to lakes and wetlands affected by eutrophication in that eutrophic water bodies have improved sources of food for wild birds (Pretty et al., 2002; Finnish Environmental Institute, 2005).

2.4 Methods for estimating the costs and benefits of eutrophication

Environmental valuation serves to value environmental goods and services in monetary terms so they can be considered in decision-making and management. Environmental goods and services such as clean air and water, clean beaches, attractive landscapes, wilderness and wildlife cannot be valued as conventional goods and services. Their value, and the costs of any pollution or degradation of these, is therefore, often ignored. The high value of water-frontage property, the cost of visits to see spectacular scenery, the damaging cost of oil spills on wildlife and aquatic ecosystems, and the cost to users of salt in water are some of the values which cannot be traded in the market place (Lothian, 1999).

Market-based approaches may be used to estimate water treatment costs. For example, Dennison and Lyne (1997) identified the main contaminants responsible for high treatment costs at two water treatment plants in the Umgeni catchment of KZN and used regression analysis to determine the relationship between observed water contaminant levels and observed water treatment costs. The estimated relationship was then used to predict treatment costs for various levels of contaminants. A similar approach was adopted in this research work by Gebremedhin (2009).

Several techniques are used under surrogate market methods, the most popular being hedonic pricing and travel cost. Batalhone et al. (2002) point out that in the housing market individuals make decisions based on environmental and location characteristics, i.e. hedonic methods. The price of the house is related to characteristics of the environment in which it is situated. Therefore, the value of the water quality or other environmental good can be deduced from the price of the house. The cost of reducing eutrophication can be estimated from the price of the house in the surrogate market. Gibbs (2002) also used water clarity as a measure of the degree of eutrophication levels.

The Travel Cost Method is typically used to estimate the use-value of a recreation site by using the costs (e.g. of time and travel) incurred by visitors to the recreation site to estimate the demand for recreation at that site. For example, Soutukorva (2005) estimated the benefits from reduced eutrophication in the Stockholm archipelago of Sweden.

When the value of environmental quality cannot be estimated by market information, the contingent valuation method (CVM) is one of the best methods that can be applied in estimating the cost of “water quality” associated with reducing eutrophication. CVM is a survey-based economic technique for the valuation of non-market resources, based on the idea of knowing people’s willingness-to-pay for certain characteristics of their environment by asking them directly. Zylicz et al. (1995) estimated the economic value of eutrophication damage in the Baltic Sea region by applying this approach.

2.4.1 Water quality costs

Measurement of the benefits and costs of an improvement in water quality are often difficult, with two principal reasons for this problem. First, for a complete analysis, all relevant benefits and costs have to be measured, i.e. the effects on all parties concerned have to be taken into account. Second, the physical benefits and costs have to be measured in monetary terms (UNEP, 2005).

A number of tools exist to rank and prioritise environmental problems using monetary valuation methods. Broadly, these valuation techniques are grouped into two types of markets, namely implicit markets and constructed markets (FAO, 2002). If markets (implicit markets) existed for environmental goods and services, their economic value would be the sum of actual payments for the commodities plus an appropriate measure of consumer surplus. Consumer surplus generally refers to the excess of individual's willingness-to-pay (WTP) for a good as reflected in a demand curve, over actual payment. However, measurements of the benefits and costs of environmental improvements are difficult because typically there are no markets for environmental quality. Nevertheless, there are some characteristics in markets that relate to environmental quality, and it is sometimes possible to measure peoples' willingness-to-pay for the environmental goods and services by using data from those markets (Braden and Kolstad, 1991). Hence, there are implicit market-based methods for estimating costs and benefits of an improvement in water quality such as hedonic methods and travel cost methods.

In an implicit market, prices are obtained by observing the market prices of goods and services strictly related to the good or service to be valued. For example, the cost of air or water pollution can be estimated by calculating the impacts of pollution on the market prices of houses. However, it is not always possible to value environmental goods in this way since there are other values associated with environmental goods, such as aesthetic values and species diversity, which are not connected to markets. These values are often measured through constructed (hypothetical) markets. The WTP expressed in constructed (hypothetical) markets is a measure of the value people place on the particular good (UNEP, 2005). In a constructed market, prices are obtained through questionnaires, which simulate a hypothetical market for the good or service (FAO, 2002).

2.4.1.1 Implicit (surrogate) market based methods

Market based methods can be used when people make choices in the market place among goods and services that have some characteristics related to the environment (UNEP, 2005). The benefits of protection or remediation of eutrophication cannot always be estimated easily. One of the most popular market based valuation methods is the hedonic pricing method.

2.4.1.1.1 Hedonic pricing (HP)

The basic idea of the hedonic pricing method is that when an individual decides to buy a residence, he or she takes his/her decision based on environmental and location characteristics (Batalhone et al., 2002). Batalhone et al. (2002) indicate that there is a considerable reduction in property market values due to the presence of a "poor" environment. When the value of a good is related to the many characteristics it possesses, these relationships can be studied for price differences to deduce the value people place on one of those characteristics (Field, 1997). The property market is of use in environmental economics as houses differ in the environmental quality at their location (Braden and Kolstad, 1991). Therefore, the price of environmental characteristics can be derived from the differences in house prices.

Property value models

Waterside property owners are potentially the recipients of the greatest economic gains from improved water quality because the benefits of water quality can be capitalised in the price of the waterside property (Michael et al., 1996). Hence, the impacts of water pollution (eutrophication) can be valued from the property prices of waterside properties. Suppose there are two houses that are the same in terms of all their physical characteristics, as well as in location factors. However, if one of the houses is located in an area of substantially eutrophic water, and the other is located in an area of relatively clean water, it is expected that the market price of these two houses will differ as a result of the eutrophication. Gibbs (2002) used water clarity as a measure of the degree of eutrophication and showed that water clarity (and hence eutrophication) had a significant effect on the prices paid for residential properties.

Both the studies by Michael et al. (1996) and Gibbs (2002) assume a decreasing willingness-to-pay per unit of improvement as water quality improves. Property owners who live on a lake/river with visibility to only one meter would be willing to pay more for a one meter improvement in clarity than owners who live on lakes/rivers with water that is already clear to four meters (UNEP, 2005). Waterside properties differ in many respects, not just in terms of the water quality. Hence, it is necessary to collect large amounts of data on many such properties and to apply statistical techniques to identify the relationship between each characteristic and the property value (UNEP, 2005).

In examining the difference in value between two water bodies D'Arge and Shogren (1989) used three different valuation techniques:

- A site valuation based on comparing property values between the two water bodies (lakes).
- A market value approach based on asking real estate agents in the area to identify reasons for the observed price differential.
- A contingent valuation approach asking residents their willingness-to-pay for improved water quality.

Pretty et al. (2002) advocate that in order to calculate the potential effects of eutrophication on land values, data is needed on the length of rivers affected by eutrophication, and the number of properties affected. The value-loss relationship according to Pretty et al. (2002) is:

$VL_{A2} = P_n \text{ x } F_c \text{ x } V_{Lp}$

Where:
VL_{A2} = total value-loss for waterside properties in England and Wales;
P_n = number of waterside properties;
F_c = frequency of loss of value due to some eutrophication; and
V_{Lp} = value-loss per average 10 metre frontage.

Poor et al. (2007) explore the effect of ambient water quality on waterfront-housing prices in the St. Mary's River watershed, U.S.A. They use the semi-logarithmic functional form to estimate the hedonic price function where the natural logarithm of the real sale price (LNREALPR) for each property (i) is the dependent variable which is regressed on a set of structural characteristics (S), neighbourhood characteristics (N), and environmental water quality characteristics (E). Poor et al. (2007) estimate the regression model as follows:

$$\text{LNREALPR}_i = \alpha + \beta_{1i}S + \beta_{2i}N + \beta_{3i}E + \varepsilon_i$$

Where α, β_1, β_2, β_3, are the coefficients to be estimated and ε_i is a random error term. The structural characteristics include: a garage dummy variable, the number of stories, a porch/deck area variable, and number of fireplaces. Neighbourhood characteristics include: distance to the primary employer and largest town within the watershed, which are expected to have a positive impact on property values. The water quality measures included in the model are yearly averages of dissolved inorganic nitrogen and total suspended solids, which are expected to have a negative impact on sales prices (Poor et al., 2007).

Boyle et al. (1999) estimate hedonic price functions relating the price of the property to the characteristics of the property, its structure, and its location. They estimate, for each market area:

$$\text{PP} = \sum_j \beta_j A_j + \beta_{WC}\,\text{LKAREA} * \ln(\text{WC})$$

Where:

PP = property's sales price;

B = coefficients;

A_j = vector of property characteristics that are assumed to affect sales prices (e.g. square area of living area, distance to nearest city, etc.);

LKAREA = total surface area of the lake on which the property is located; and

ln (WC) = natural log of the minimum water clarity of the lakes during the summer months of the year in which the property was purchased.

Wage differential approach

Another hedonic pricing method is the wage differential approach. This approach uses information regarding the difference in remuneration for jobs with different degrees of health risk. This approach is used to place values on morbidity and mortality risks. The approach is based on the theory that workers have to be paid a premium to undertake jobs that are naturally risky and this information can be used to estimate the implicit value individuals place on sickness or premature death. In other words, a higher wage would be needed to induce people to work in polluted areas or to undertake risky jobs (Dixon et al., 1994; Field, 1997).

It is unlikely that the wage differential approach is relevant to estimating the cost of water pollution unless the levels of pollution pose a danger to the health of employees. This is unlikely in the case of the Vaal River, although algal toxins and associated health risks may raise this possibility in the future as more knowledge is gained about the risks associated with exposure to the toxins.

2.4.1.1.2 Travel Cost Method (TCM)

The travel cost method usually uses market information to estimate the value of recreation sites. Visitors to such sites incur time and travel expenses which are a proxy for price and reflect the willingness-to-pay for the characteristics of the site (UNEP, 2005). This method uses data on the type and number of recreation trips that people make to different sites at varying levels of expenses.

Soutukorva (2005) applied this method and estimated the benefits from reduced eutrophication using sight depth records and data from large-scale questionnaire surveys on travel behaviour and visits in the Stockholm archipelago of Sweden. One underlying assumption behind the TCM is that a consumer must visit a recreational site in order to consume its services. The data for the TCM is provided by survey examination of actual or prospective visitors of a recreational site, i.e. only use-values are considered. The fact that only use-values are considered in the TCM follows from the weak complimentary assumption, i.e. if the individual does not consume the environmental commodity his/her utility is unaffected by changes in the quality of the commodity.

Value-loss costs are incurred when people are prevented by eutrophication and algal blooms from enjoying the quality of a water body. In order to calculate the costs of eutrophication on recreation, Pretty et al. (2002) used values per person per year to estimate the benefit derived from water courses by visitors in the UK. This is termed the consumer surplus, individual willingness-to-pay, and according to Pretty et al. (2002) is mostly in the range of £8-20 per person per visit.

The value-loss relationship for recreation is described by Pretty et al. (2002) as:

$VL_{A5} = N_v \times F_c \times C_s$

Where:

VL_{A5} = reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling and general amenity (picnics, walking, aesthetics);
N_v = number of day and tourist-day visits to water bodies made each year;
F_c = frequency of closure (% of days); and
C_s = consumer surplus (£ per day) for use of water-based ecosystem services.

Additionally contingent valuation methods (CVM) and conjoint analysis could use hypothetical scenarios to elicit the impact of, for example, water quality and fishing quality, on tourists' demand for recreation activities at a particular site.

Hanley (1989) applies CVM and the TCM to the problem of valuing non-market recreation benefits derived by visitors to a part of the Queen Elizabeth Forest Park in Central Scotland. The purpose of the contingent valuation exercise was to place a monetary value on the benefits derived by visitors from certain aspects of the Forest Park such as wildlife, landscape and recreational facilities. For each of these Hanley (1989) constructed a hypothetical market. Hanley (1989) also made use of the TCM to infer the value that consumers place on a non-marketed good by observing their behaviour in actual markets. Their minimum willingness-to-pay to consume the services provided by the Forest Park can be measured by estimating their travelling costs, plus any other costs they incur in consuming these services.

2.4.1.2 Constructed market based methods

The value of environmental quality cannot always be estimated from market information. In such cases, methods based on constructed (hypothetical) markets are often used (UNEP, 2005). These methods typically involve survey research to solicit verbal willingness-to-pay for specified environmental amenities.

Contingent valuation methods are useful in the absence of price information and competitive markets and are direct approaches to estimating willingness-to-pay (Field, 1997). The CVM is based on the simple idea of knowing peoples' willingness-to-pay for certain characteristics of their environment by asking them.

Numerous contingent valuation studies have been done to date, for a number of environmental factors. Zylicz et al. (1995) established the CVM to estimate the economic value of eutrophication damage in the Baltic Sea region. Their study explored the willingness-to-pay of beachgoers to clean up the Baltic Sea and coast. A contingent market encompasses the good itself, the institutional context in which it would be provided, and the way it would be financed (Zylicz et al., 1995). According to Field (1997), there are four steps that should be followed in CV analysis, i.e.

- Identification and description of the environmental characteristics to be evaluated.
- Identification of respondents to be approached, including sampling procedures used to select respondents.
- Design and application of a survey questionnaire through personal, phone or mail interviews (in recent years focus groups have been used).
- Analysis of results and aggregation of individual responses to estimate values for the group affected by the environmental change.

The major criticism of the CVM is the possibility of inaccurate results due to the method's hypothetical nature (UNEP, 2005). Mirrilees et al. (2003) quantified the value of water resources of the Vaal river system and compared them with full economic costs. The accuracy of the results, compared against the real world, was inconsistent. Due to differences in the type of water use in the Vaal river system, they divided the river into two sections; the upper Vaal and the middle Vaal. The water use of the upper Vaal is mainly industrial (including mining) and domestic and that of the middle Vaal largely agricultural and mining.

The main water sectors are:

- municipal use (subdivided into household, light industry and parks),
- irrigation use,
- afforestation use,
- electricity use, and
- heavy industry use (including mining).

Dennison & Lyne (1997) investigated water quality and treatment costs in the Umgeni River catchment area. In their study, they used models of water treatment costs for two different treatment plants. They were able to identify the main contaminants responsible for raw water treatment costs in the Umgeni catchment area, and then predicted treatment costs from the observed levels of contaminants. A partial adjustments model, using ordinary least squares regression and principal component analysis, was the approach they applied in their research. Their study established that environmental and biological contaminants have a marked impact on real water treatment costs.

2.4.2 Economic sector costs

A summary of the different costs of eutrophication and possible methodologies of estimating these costs are summarised in Table 2.2 along with an indication of the methodologies adopted for use in this study.

2.4.2.1 Agricultural model

Pretty et al. (2002) illustrated the costs of eutrophication on industry as a whole. Included in this sector is livestock watering and irrigation which both fall under the agricultural sector. It was, therefore, deemed relevant that this equation be included under the section on agriculture. The value-loss relationship is estimated as follows:

$$VL = V_w \times F_c$$

Where:

VL = reduced value of water bodies for abstraction, livestock watering, navigation, irrigation and industrial uses;

V_w = value of water for industrial, farming and navigation uses; and

F_c = frequency of closure (prevention of use of water for demand use).

Table 2.2: Possible methods of quantifying the economic costs of eutrophication and the methods applied in this study.

Cost category	Possible estimating methodology	Methodology used in this study
Damage costs arising from eutrophication		
Social damage costs: use value		
• Reduced value of water side property	Hedonic models	Seemingly unrelated regression based on hedonic price models
• Health costs for humans & reduced value of water bodies for livestock	Willingness-to-pay, wage differential (hedonic pricing methods)	Fixed effects regression
• Drinking water treatment costs	Partial adjustment regression model, ordinary least squares regression model	Ordinary least squares regression, partial adjustment regression and laboratory tests
• Clean up costs of water ways	Ordinary least squares regression	Not considered in this study
• Reduced recreational & amenity value of water bodies	Contingent valuation, travel cost & hedonic methods	Contingent valuation method and conjoint analysis
• Fisheries	Direct assessment	Not considered in this study
Ecological damage costs: non-use value	Contingent valuation method & willingness-to-pay indirect method	Not considered in this study
Costs of addressing eutrophication	Direct market prices	Not considered in this study
Research and development costs of eutrophication	Direct market prices	Not considered in this study

2.4.2.2 Industry model

The same equation employed for the agricultural model is applicable to the industry sector. Similarly the value-loss relationship is estimated as follows:

$VL = V_w \times F_c$

Where:

VL = reduced value of water bodies for abstraction, livestock watering, navigation, irrigation and industrial uses;

V_w = value of water for industrial, farming and navigation uses; and

F_c = frequency of closure (prevention of use of water for demand use).

2.4.2.3 Fishery model

Pretty et al. (2002) described the value-loss relationship for fisheries as:

$VL = V_f \times F_c$

Where:

VL = revenue losses for commercial freshwater aquaculture and fisheries;

V_f = value of commercial inland and shell-fisheries in the UK; and

F_c = frequency of closure (damage to fishery).

2.4.2.4 Tourism model

The value-loss relationship for tourism is illustrated as:

$VLTotal = N_v \times F_c \times Eday$

Where:

VL = revenue losses for formal tourist industry (inland and coastal);

N_v = number of day and tourist-day visits to water bodies made each year;

F_c = frequency of closure (% of days); and

$Eday$ = total expenditure per day and tourist-day visit.

2.4.2.5 Ecosystem model

The value-loss costs related to the intrinsic value (non-use) of species and ecosystems affected by eutrophication are difficult to measure (Pretty et al., 2002). Pretty et al. (2002) make extensive use of the UK National Biodiversity and Habitat Action Plans; which list several important water species and habitats that are adversely affected by eutrophication. These plans contain costed targets and action plans for 406 species and 38 key habitats in the UK. They use the cost of restoring these species and habitats as a proxy for the eutrophication costs.

For those species and habitats for which eutrophication is identified as one of the factors causing problems, Pretty et al. (2002) have used details in the Biodiversity Action Plans (BAP's), Habitat Action Plans (HAP's) and Species Action Plans (SAP's) to estimate the costs.

Pretty et al. (2002) described the relationship for the value-loss as:

$$VL = C_e + C_m + (S \text{ x } C_s \text{ x } P)$$

Where:

VL = negative ecological effects on biota resulting in changed species composition (biodiversity) and loss of key or sensitive species;

C_e = average annual cost of HAP's addressing eutrophic lakes;

C_m = average annual cost of HAP's addressing mesotrophic lakes;

S = number of Species Action Plans potentially affected by eutrophication;

C_s = average annual cost of SAP's; and

P = proportion of SAP's affected by eutrophication;

Further, CVM's can be used to assess the existence value of natural biodiversity in a river system.

2.4.2.6 Clean-up costs of water bodies model

The damage cost relationship for clean-up costs, according to Pretty et al. (2002), is:

$DC = (\Sigma W_{ci\text{-}j}) \times P$

Where:

DC = clean-up costs of waterways (dredging, weed-cutting);

ΣW_c = sum of costs of weed cutting for organisations i to j due to eutrophication; and

P = proportion of weed cutting that can be attributed to eutrophication.

2.4.3 Direct costs of water treatment

Drinking water treatment is the process of removing contaminants from surface or ground water to make it safe and palatable for human consumption. The cost of water treatment may be influenced by factors such as: water consumption (influenced by composition of customers), age of system and size of utility, technology or process involved, energy costs, and labour costs (Sauer and Kimber, 2002).

Users of clean water are categorised into three groups namely domestic, industrial and commercial (DWAF, 2001). As each sector's water quality requirements differ (Karamouz et al., 2003) it is important to assess the costs of water treatment separately for each sector.

2.4.3.1 Treatment costs for potable water

Potable water treatment encompasses all processes and technologies that are used to remove turbidity, chemical pollutants and microorganisms from water, to a standard fit for human consumption (WRC, 2002). Potable water is a specific and specially treated kind of water. The quality of the source water determines what treatment process will be suitable (Msibi, 2002). Pretty et al. (2002) divide water treatment into two categories: removal of algal toxins and algal decomposition products, and costs to remove nitrogen.

Removal of algal toxins

The damage cost relationship for this category, according to Pretty et al. (2002), is:

$DC_{A8} = (C_o \times A_p \times ASP_o) + (C_c \times A_p \times ASP_c) + C_r$

Where:

DC_{A8} = drinking water treatment costs (treatments and actions to remove toxins and algal decomposition products);

C_o = annual operating expenditure by water companies;

C_c = annual capital expenditure by water companies;

A_p = proportion of production liable to suffer from algal proliferation;

ASP_o = proportion of Algae Sensitive Production (ASP) operating costs for eutrophication;

ASP_c = proportion of ASP capital costs for eutrophication; and

C_r = annual cost of reservoir management systems.

Removal of Nitrogen

Pretty et al. (2002) described the damage costs for the removal of nitrogen as:

$$CC_{A9} = NC_o + NC_c$$

Where:

CC_{A9} = drinking water treatment costs (to remove nitrate);

NC_o = annual operating costs of removal of nitrate by water companies; and

NC_c = annual capital costs of removal of nitrate by water companies.

2.4.3.2 Treatment costs for industry

Non-potable water is primarily used in industry, commerce and irrigation where the quality does not have to be as high as for potable purposes. Industrial water use for the most part is low-cost and self supplied. Therefore, it is expected to have different costs and treatment requirements compared to potable water (Drinan, 2000). The Vaal River system is of great importance as an industrial water source for industries such as Sasol and Eskom in Mpumalanga province through the transfer of water from the Vaal River system (DWA, 2011).

2.4.4 Costs of monitoring and treating pollution

Monitoring pollution levels, identifying polluters, determining impact costs, collecting discharge information, sending out accounts, receiving payments, and recording data and transactions of eutrophication involves significant costs. These costs are often administered by government institutions such as the Department of Water Affairs and Forestry (DWAF) (Taviv et al., 1999). According to Taviv et al. (1999) monitoring has four objectives:

- To monitor whether charge systems are having any impact on water quality;

- To establish whether the pollution loads disclosed on a voluntary basis by the polluters agree with independently monitored data by carrying out spot checks on individual polluters;
- To monitor streams at control points;
- To monitor the background contribution and loads transferred from other catchments.

Some studies reviewed emphasised that monitoring costs of pollution can be minimised effectively by relying on self-monitoring by enterprises combined with sufficient random checks to ensure that enterprises operate their monitoring systems properly. Self-monitoring is also an important mechanism for creating environmental awareness among senior managers of polluting enterprises or any economic agent (Ackermann, 1997).

Pretty et al. (2002) separated this cost category into monitoring costs and compliance control costs. Compliance costs were further divided into sewerage treatment costs and the cost for the treatment of algal blooms.

2.4.4.1 Monitoring costs

Statutory agencies incur costs to monitor water bodies for the presence of algae and algal decomposition products. The monitoring costs are described by Pretty et al. (2002) as:

$$MC_{A11} = \Sigma M_{ci\text{-}j}$$

Where:
MC_{A11} = monitoring costs for water; and
M_c = monitoring costs for organisations i to j.

2.4.4.2 Compliance costs

Compliance control costs form part of the costs of monitoring and treating pollution and include sewage treatment costs (i.e. reducing nutrient concentrations in effluent discharges), costs of treatment of alga blooms, and the cost of adopting new farming practices to emit fewer nutrients.

Treatment of algal blooms

Pretty et al. (2002) described the damage cost relationship for this category as:

$DC_{A12} = \Sigma C_{ti-j}$

Where:

DC_{A12} = cost of treatment of algal blooms and preventative measures; and

C_t = sum of treatment costs by water companies i to j.

Sewerage treatment costs

The compliance (damage) costs for this category, according to Pretty et al. (2002), are:

$CC_{A13} = PC_o + PC_c$

Where:

CC_{A13}= sewage treatment costs to remove phosphate;

PC_o = annual operating costs of removal of phosphate by water companies; and

PC_c = annual capital costs of removal of phosphate by water companies.

3. METHODOLOGY

3.1 Study area

The Vaal River is 1300 kilometres long and stretches from the Drakensberg plateau to the arid Karoo region. The Vaal River is a major river system in South Africa and is the primary supplier of water to the economic heartland of South Africa. The principal use of water from the Vaal River system is for domestic (including agriculture) and industrial purposes (including mining and for cooling electricity power stations).The Vaal River system was separated into three parts for the purpose of this study: the Grootdraai Dam situated in the upper area of the Vaal River, the Vaal Dam situated in middle of the Vaal River, and Bloemhof Dam situated in the lower reaches of the Vaal River. See Figure 3.1 for a map of the Vaal River catchment area.

Figure 3.1: The Vaal River catchment area.
Source: DWAF (2002)

The Grootdraai Dam is situated 10 kilometres from the town of Standerton in the Mpumalanga province of South Africa. Grootdraai Dam was completed in 1982 and has a full supply capacity of 364 million cubic metres. The Grootdraai Dam primarily supplies water to municipal users and industrial users located in the Secunda area of the Mpumalanga province of South Africa. The Vaal Dam is located 56 kilometres south of the city of Johannesburg in the Gauteng Province of South Africa. The Vaal Dam was completed in 1938 and at full supply capacity can store up to 2536 million cubic metres. The Vaal Dam is critical to the water supply infrastructure of Gauteng province and to other surrounding provinces and supplies water to municipal users and industrial users. The Bloemhof Dam is situated two kilometres from the town of Bloemhof in the North West province of South Africa. The Bloemhof Dam was completed in 1970 and has a full supply capacity of 1269 million cubic metres. The Bloemhof Dam supplies water to the lower Vaal River area for industrial, municipal and agricultural users (Sibande, 2010).

3.2 Model specification and data description by economic sector

The models used to estimate the costs associated with eutrophication for the different sectors were based on regression analysis. Gujarati (2003:18) described regression analysis as "... the study of the dependence of one variable, the *dependent variable,* on one or more other variables, the *explanatory variables,* with a view to estimating and/or predicting the (population) mean or average value of the former in terms of the known or fixed (in repeated sampling) values of the latter." For each economic sector considered in the study, the dependent variable was the relevant sector's total costs (e.g. total agricultural costs) and the explanatory variables were those variables expected to explain total costs including variables representing eutrophication (e.g. the concentration of ammonia present in the water). Data collected for the variables and the specified models were used to estimate a relationship between the dependent variable and the explanatory variables. This estimated relationship describes the effect of a change in the explanatory variable on the total cost variable and can be used to ascertain the impact of eutrophication on the particular sector. The estimated relationship between total costs to the sector and the eutrophication variables could then be used to evaluate the extent of costs associated with eutrophication for the sector. The following sections describe the data and model specification for each of the economic sectors considered in this study.

3.2.1 Agricultural cost model

Agricultural cost data were obtained from Grain South Africa (www.grainsa.co.za/) and were grouped into capital costs and intermediate costs. Table 3.1 indicates the costs included in each of the groups. All other costs (e.g. fertiliser) were excluded so that variables possibly affected by eutrophication were not included as independent variables. All the costs listed in Table 3.1 are secondary data (not collected by the researcher) and although they are crop specific, are not influenced by water quality. The agricultural cost data were pooled (agricultural cost data for the three areas were combined).

Table 3.1: Capital and intermediate agricultural costs included in the agricultural cost model for the Vaal catchment area.

Capital costs	Intermediate costs
Implement costs	Seed costs
Machinery costs	Fuel costs
Financing costs	Labour costs
Irrigation equipment costs	Insurance costs
	Packing material costs
	Maintenance and repair costs

Source: de Villiers (2009)

A fixed effects model was generated to estimate the costs associated with eutrophication for the agricultural sector in the Vaal River catchment. The fixed effects model permits cross-section heterogeneity in the error term, allowing for the cross-sections to differ in terms of the intercept term (even though the data for three cross-sections/dams have been pooled).The agricultural cost data were log-linearised to represent percentage changes in the variables and the relationship between the changes.

Mathematically, the change in total agricultural cost is a function of the changes in capital costs, intermediate costs and eutrophication levels.

The fixed effects model was specified as:

$$\text{LN\_COST\_LOW} = B_1 + B_2\text{LN\_INT\_LOW} + B_3\text{LN\_CAP\_LOW} + B_4\text{LN\_P\_LOW} + B_5\text{LN\_N\_LOW}$$

Where:

LN_COST_LOW is the natural log of total agricultural costs for the lower Vaal;

LN_INT_LOW is the natural log of intermediate agricultural costs for the lower Vaal;

LN_CAP_LOW is the natural log of capital agricultural costs for the lower Vaal;
LN_P_LOW is the natural log of the phosphorus level for the lower Vaal;
LN_N_LOW is the natural log of the nitrogen level for the lower Vaal; and
B_1 to B_5 are coefficients to be estimated.

Similar models were specified for the middle (MID) and upper (UP) Vaal areas.

The total cost incurred by farmers over time was established for all major crop groups as well as for different areas along the Vaal River. However, the data were not sufficient to build individual models for each crop group for each area. Only the 2003 rand values for capital and intermediate costs were available. Therefore, price indices were used to calculate what the change from 2003 was for all the other years in relation to this value. The weights for the indices were determined according to Grain SA indices data. Therefore, total costs faced over time were converted into indices rather than magnitudes. This resulted in a single time series made up of weighted aggregated indices for each geographical area. The index was used as the dependent variable in the model to estimate changes in agricultural costs due to eutrophication. All costs were converted to cost/ha.

3.2.2 Industry cost model

In collecting data concerning the use of water by industries in the study area, an identity problem arose as many of the users registered as industries in the DWAF registered water users report were actually farmers. Eskom and Sasol were the largest, non-agriculture water users registered in the industry category and, therefore, emphasis was placed on data relating to them and expanded to the industry sector as a whole. Discussions were held with Eskom and Sasol to gauge the extent to which eutrophication affected their production. Data collected from these discussions did not allow for econometric analysis. For a fuller explanation and rationale for this see de Villiers (2009, Section 4.3.2:36). It was, however, established that water quality in general and not eutrophication *per se*, is the dominant concern for industries such as Eskom and Sasol as the treatment process for salt indirectly addresses the symptoms of eutrophication.

Should eutrophication increase to untreatable levels for such industries, they have the option of obtaining clean water from a water supplier, such as Rand Water. For the mining industry, the willingness to pay for clean water, or the cost of eutrophication, will be equivalent to the current tariff charged by water suppliers. Hence, the water treatment costs incurred by water supply services are relevant to this study as they indirectly affect domestic, agriculture and industry water users through the tariffs charged to consumers.

It was decided to rather use data regarding the costs incurred by water supply services due to eutrophication and to transfer these costs back to mines and industries through the tariffs they paid for the water consumed. This is an indirect way of using water supply services to estimate the cost to industry of eutrophication.

3.2.3 Water treatment costs

Two studies within the project investigated the costs of treating water, specifically the impact of eutrophication on these costs. The first study used the costs incurred by water supply services (Rand Water) due to eutrophication as an indirect indication of the costs to industry of eutrophication (de Villiers, 2009). The second study investigated the relationship between raw water quality and the chemical costs of producing potable water at two water treatment plants: Zuikerbosch Station #2 (owned by Rand Water) in the Upper Vaal Water Management Area and Balkfontein (owned by Sedibeng Water) in the Middle Vaal Water Management Area (Gebremedhin, 2009).

3.2.3.1 Costs incurred by water supply services

Water treatment cost data was sourced from Rand Water. Rand Water is the biggest bulk water processor in the Vaal River System. Data were available for the entire study period upon request. Dam-specific data were not available and, therefore, data from the Vaal Dam were used as a proxy for the Bloemhof and Grootdraai Dams (Rand Water primarily processes raw water from the Vaal Dam). The data gathered from Rand Water included the total cost and input costs of treating water over time, tariffs charged for water over time, the cost of buying raw water, and the total revenue earned by selling water. The water treatment costs were transformed into cost per kilolitre. Mathematically, the change in treatment costs per kilolitre of water is a function of the change in eutrophication level.

Ordinary least squares (OLS) regression was used to estimate the following model:

$$\mathrm{LN\_C\_L} = B_1 + B_2\mathrm{LN\_N} + B_3\mathrm{LN\_P}$$

Where:

LN_C_L is the natural log of the cost per kilolitre to treat water;

LN_N is the natural log of the nitrogen level;

LN_P is the natural log of the phosphorus level; and

B_1 to B_3 are coefficients to be estimated.

The eutrophication variables included in the above model are the same as used in the agricultural models.

3.2.3.2 The relationship between raw water quality and the chemical costs of producing potable water

Time series data on raw water quality and chemical dosages used to treat raw water were obtained for Zuikerbosch Station #2 (hereafter referred to as Zuikerbosch) for the period November 2004 to October 2006 and for Balkfontein for the period January 2004 to December 2006. Zuikerbosch and Balkfontein water treatment plants were studied because they were identified as important, but geographically separate, water treatment stations along the Vaal River.

Following Dennison and Lyne (1997), Graham et al. (1998), Forster et al. (1987) and Dearmont et al. (1998), this study used multiple regression analysis to relate the costs of producing potable water from raw water to characteristics of the raw water. For both water treatment plants, ordinary least squares (OLS) regression was used to identify the relationship between real chemical costs of water treatment and the dimensions of water quality identified through the respective principal component analyses. For predictive, rather than explanatory purposes, a partial adjustment regression model was estimated for each of the two water treatment plants. Using this model, real chemical water treatment costs were specified as a function of real chemical water treatment costs in the previous time period and of raw water quality variables in the current period. For a full discussion of the methodology used in this component of the study see Gebremedhin (2009).

The partial adjustment model for the Zuikerbosch treatment plant was specified as:

$$(\text{Real chm cost})_t = B_0 + B_1 (NO_3)_t + B_2 (\text{Total\_alkalinity})_t + B_3 EC_t + B_4 (SO_4)_t + B_5 (NO_3)_t^2 + B_6 (TA)_t^2 + B_7 (SO_4)_t^2 + B_8 (\text{H Chl 665})_t + B_9 (NTU)_t + B_{10} (\text{Real chm cost})_{t-1}$$

Where:

$(\text{Real chm cost})_t$ is the real water treatment cost per ML (Rand) in time period t;

$(NO_3)_t$ is the nitrate loading (mg/L) in time period t;

$(\text{Total\_alkalinity})_t$ is the total alkalinity loading (mg/L) in time period t;

$(EC)_t$ is electrical conductivity (dS/m) in time period t;

$(SO_4)_t$ is sulphate loading (mg/L) in time period t;

$(\text{H Chl 665})_t$ is chlorophyll (total photosynthetic pigments) (μg/L) in time period t;

$(NTU)_t$ is turbidity (NTU) in time period *t*; and

B_i are estimated regression coefficients (i = 1, 2, …, 10).

Similarly, the algebraic model for the Balkfontein water treatment plant was specified as:

$$(\text{Real chm cost})_t = B_0 + B_1 (\text{Chl-a})_t + B_2 (\text{Turb})_t + B_3 (\text{Colour})_t + B_4 (\text{Temp})_t + B_5 (Cl)_t + B_6 (Ca)_t + B_7 (Fe)_t + B_8 (Mn)_t + B_9 (\text{Thard})_t + B_{10} (Ca^2)_t + B_{11} (Mn^2)_t + B_{12} (Cl^2)_t + B_{13} (\text{Real chm cost})_{t-1}$$

Where:

$(\text{Real chm cost})_t$ is the real water treatment cost per ML (Rand) in time period t;

$(\text{Chl-a})_t$ is chlorophyll ‘a’ (μg/L);

$(\text{Turb})_t$ is turbidity (NTU: Nephelometric Turbidity Units);

$(\text{Colour})_t$ is colour (Pt-co: Platinum Cobalt Standard);

$(\text{Temp})_t$ is temperature (degree Celsius);

$(Cl)_t$ is chloride (mg/L);

$(Ca)_t$ is calcium (mg/L);

$(Fe)_t$ is iron (μg/L);

$(Mn)_t$ is manganese (μg/L);

$(\text{Thard})_t$ is total hardness (mg/L);

$(\text{Real chm cost})_{t-1}$ is real water treatment cost lagged by one week(R); and

B_i are estimated regression coefficients (i = 1, 2, …, 13).

3.2.4 Property price model

Secondary property price data for all three study sites were obtained from the Knowledge Factory (www.knowledgefactory.co.za/), a customer insights company which is part of the Primedia group. The prices were adjusted for inflation using the ABSA Bank house price index.

A seemingly unrelated regression (SUR) model was used to estimate the relationship between property prices and eutrophication for all three study sites. A SUR model includes several individual or underlying relationships, linked by the fact that their disturbances are correlated (Moon and Perron, 2006). Moon et al. (2006) motivates two reasons for the use of SUR models: (1) to increase the efficiency of the regression model by incorporating information from different equations; and (2) to impose and test certain restrictions involving parameters in these different equations. The SUR model is a pooled model, which allows multiple groups to be incorporated into one model (Baltagi, 2008). Thus, the effect of eutrophication on unit price for each geographical area can be included in the one model.

Eutrophication was proxied using data on the levels of chlorophyll 'a', ammonia and nitrates and nitrites present in the water obtained from the Department of Water Affairs and Forestry (now the Department of Water Affairs). The model is specified as follows:

$$UnitP_B = B_1 + B_2 LN\_CHLA\_B + B_3 LN\_ NH_4\_B + B_4\ LN\_NO_2NO_3\_B$$
$$UnitP_V = B_1 + B_2 LN\_CHLA\_V + B_3 LN\_ NH_4\_V + B_4\ LN\_NO_2NO_3\_V$$
$$UnitP_G = B_1 + B_2 LN\_CHLA\_G + B_3 LN\_ NH_4\_G + B_4\ LN\_NO_2NO_3\_G$$

Where:

UnitP is the natural log of the sales price of property, divided by the stand size;

LN_ CHLA is the natural log of the level of chlorophyll 'a' present in the water;

LN_ NH_4 is the natural log of the level of ammonia present in the water;

LN_NO_2NO_3 is the natural log of the level of nitrate and nitrite present in the water;

B_1 to B_4 are coefficients to be estimated; and

B, V and G represent the Bloemhof, Vaal and Grootdraai Dams respectively.

Two additional property models were constructed as above, but with a one year and two year lag of the independent variables, respectively. For the lagged models, changes in property prices now are a function of changes in the eutrophication variables one year and two years ago. For a detailed description of the construction of the property models see Mostert (2009:37).

3.2.5 Recreation value model

Conjoint analysis was used to estimate the willingness to pay (WTP) for an improvement in the water quality of the dam in question by users of the dam. Conjoint analysis is a technique used to determine preferences of individuals across various characteristics of a multi-attribute choice (Mostert, 2009). The WTP value represents a Rand value per person, per night's stay. Three surveys were conducted in the three areas. A total of 90 respondents returned their questionnaires, with 20 respondents for Grootdraai Dam, 30 for Bloemhof Dam and 40 for the Vaal Dam (Mostert, 2009). The questionnaire responses were used to construct three value-loss relationship models; one for each geographical area. For a full discussion of the construction of the questionnaire and models see Mostert (2009:44).

The value-loss model was specified as:

VL = calculated Q as a function of R

Where:

VL = reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing, etc.), angling and general amenity (picnics, walking, aesthetics);

R = ordered number of respondents to questionnaire plus respondent characteristics; and

Q = willingness to pay.

'R' indicates the number (n) of respondents subjected to the questionnaire and the details of the characteristics of each individual respondent. 'Q' represents the individual respondents' willingness to pay for the improvement of the water body's water quality, calculated using a method suggested by the World Bank Institute's "Environmental Economics and Development Policy Course" (2002) found in session 28 (Mostert, 2009).

3.3 Integrated model specification and data description

The goal of this phase of the project was to investigate the existence of tradeoffs between the different economic costs associated with eutrophication in the Vaal River System. This was undertaken with the aim of understanding the water quality management policy implications that follow as a result of the existence of tradeoffs between the different economic costs associated with eutrophication in the Vaal River System.

Seemingly unrelated regression (SUR) was used to investigate the relationship between eutrophication and the various sectors (property, agriculture, water treatment and recreation) affected by eutrophication. The SUR econometric technique essentially allows for the interaction of the different economic costs associated with eutrophication in each of the three dams, within a single model. For example, poor water quality can affect property prices through poor quality drinking water and odour that comes from a nearby river. At the same time poor water quality can affect the agricultural sector by increasing irrigation costs through clogged irrigation pipes. Therefore, the different economic costs that are associated with eutrophication are related to some extent. Figure 3.2 illustrates that within the SUR framework, eutrophication costs that are attributable to the property, agricultural and water treatment sectors can all be specified and related within one framework.

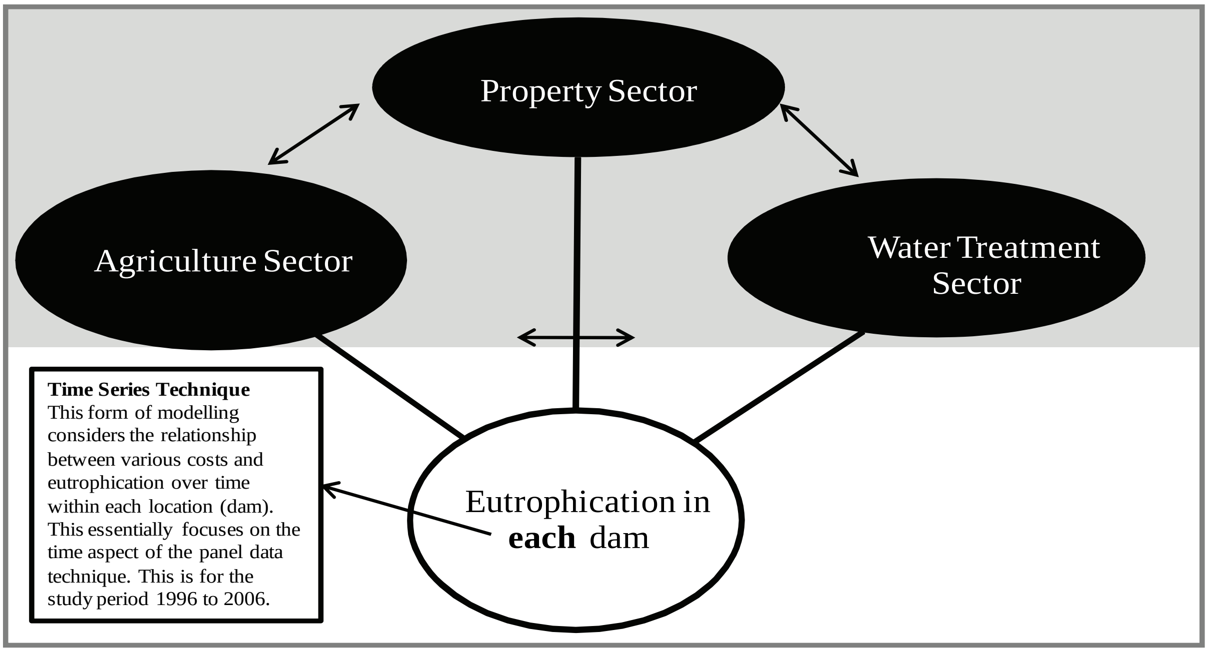

Figure 3.2: The seemingly unrelated regression framework in the context of economic costs associated with eutrophication.

This framework is specified according to time series and is the approach followed in Sibande (2010). For a more detailed discussion of the SUR technique see Sibande (2010:29).

The data used in this phase of the project were the same as those used in the agricultural and water supply service models (de Villiers, 2009) and the property and recreation models (Mostert, 2009). However, the recreational sector cost data were collected primarily at a point in time (see Mostert 2009) and were not available for the entire study period. Therefore, the analysis did not include the recreational sector due to data limitations as stated.

The SUR model was specified as follows:

Unit property price$_i$ = $p_{1i} + p_{2i}\text{Eut}_{1i} + e_{pi}$
Total agricultural cost$_i$ = $a_{1i} + a_{2i}\ \text{Eut}_i + e_{ai}$
Total water treatment cost = $i_{1i} + i_{2i}\text{Eut}_i + e_{ii}$

Where:

Unit property price = the unit property prices for property surrounding dam *i*
Total agricultural cost = the change to total cost of agriculture for a specific dam *i* due to eutrophication;
Total water treatment = the change total water treatment costs of the water supply service (Rand Water) due to eutrophication;
Eut_i = eutrophication level for specific dam *i*;
e_i = the error term for a specific dam *i*; and
p, *a* and *i* are coefficients to be estimated.
For each i = 1, 2, and 3 (Bloemhof Dam = 1; Grootdraai Dam = 2; Vaal Dam = 3)

For the property equation, eutrophication is given by the levels of Chlorophyll ‘a’, ammonia and nitrate and nitrite present in the water following Mostert (2009). For the agricultural and water treatment cost equations, eutrophication is given by the level of phosphorus and nitrogen present in the water, following de Villiers (2009). The models were estimated for the period 1996 to 2006. This period was chosen to establish a baseline for eutrophication costs in recent times.

4. ECONOMIC COSTS OF EUTROPHICATION IN THE VAAL RIVER SYSTEM: MODEL RESULTS

4.1 Results by economic sector

As discussed in Section 3.2, the models used to estimate the costs associated with eutrophication for the different sectors were based on regression analysis. In regression analysis the estimated coefficients describe the nature of the relationship between the dependent variable (e.g. agricultural costs) and a particular explanatory variable (e.g. level of phosphorus present in the water). The statistical significance of the estimated coefficient gives an indication of the strength of the relationship between the dependent variable (Y) and the explanatory variable (X). The test of significance is used to 'reject' or 'fail to reject' the null hypothesis, which is this case, is that there is no relationship between X and Y. An estimated coefficient that is significant at the one, five or 10% level implies that there is a relationship between X and Y and the null hypothesis can be rejected. The models were formulated in log linear (log) form which means the estimated value of one variable coefficient can be read directly as a percentage change, keeping all other variables constant.

4.1.1 Agricultural cost model

The agricultural cost model results as estimated by de Villiers (2009) are given in Box 1.

Box 1: Fixed effects model results for the agricultural sector, Vaal catchment area, 1982-2007

LN_COST_LOW = 0.573 + 0.739*** LN_INT_LOW +0.266***LN_CAP_LOW -
0.003LN_P_LOW + 0.006*LN_N_LOW

LN_COST_MID = 0.531 + 0.739***LN_INT_MID + 0.266***LN_CAP_MID -
0.003LN_P_MID + 0.006*LN_N_MID

LN_COST_UP = 0.549+ 0.739***LN_INT_UP + 0.266***LN_CAP_UP -
0.003LN_P_UP + 0.006*LN_N_UP

Note: LOW, MID and UP, represent the lower, middle and upper Vaal catchment areas, LN_COST is the natural log of the total agricultural costs, LN_CAP is the natural log of capital costs, LN_INT is the natural log of intermediate costs, LN_P is the natural log of the level of phosphorus present in the water and LN_N is the natural log of the level of nitrogen present in the water. *** Denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%.

Source: de Villiers (2009)

The estimated coefficient for the phosphorus level of the raw water (LN_P) in the model was not statistically significant in explaining total agricultural cost. However, the model is included here because the rejection of the inclusion of LN_P was not certain. The variable LN_N (nitrogen level of the raw water) was statistically significant in explaining variations in total agricultural cost.

In terms of capital and intermediate costs, the model behaved as expected. A change in intermediate costs such as seed and fuel costs (by 1%) would lead to a change in total cost (by 0.739%) in the same direction. This means that an increase in intermediate costs would lead to an increase in total production costs. Similarly, a 1% change in the capital cost would result in a 0.266% change in total cost in the same direction.

From the model, changes in eutrophication (P and N levels) appear to have a relatively small effect on total cost. According to the model, a 1% increase in the nitrogen level of the raw water results in a 0.006% increase in the total cost of production. Hence, the effect is relatively small, although statistically significant. As for phosphorus, a 1% increase in the phosphorus level of the raw water results in a 0.003% decrease in total cost. This is surprising, but not inexplicable. An increase in the phosphorus level of irrigation water could lead to higher crop yields and lower costs in terms of fertiliser inputs. The coefficient estimated for phosphorus was not statistically significant, suggesting that there was no relationship between agricultural costs and the level of phosphorus present in the water for the study period considered.

The relatively small effect of eutrophication on agricultural costs could be as a result of the conversion of the eutrophication data from daily observances to average annual levels. This conversion effectively smoothed out much of the variability in the eutrophication data. The ecological and economic consequences of eutrophication are most often felt as a result of the changes in the level and variability thereof at the time the variance occurs. Therefore, by smoothing out (averaging) the data, the impact of eutrophication is reduced. The results should be interpreted in the context of this limitation.

The model was re-estimated excluding the variable for phosphorus since its estimated coefficient was not statistically significant in the first model. The model results excluding the variable for phosphorus are presented in Box 2.

Box 2: Fixed effects model results for the agricultural sector, excluding phosphorus, Vaal catchment area, 1982-2007

LN_COST_LOW = 0.616 + 0.727 LN_INT_LOW + 0.275LN_CAP_LOW + 0.005LN_N_LOW
LN_COST_MID = 0.57 + 0.727LN_INT_MID + 0.275LN_CAP_MID + 0.005LN_N_MID
LN_COST_UP = 0.59 + 0.727LN_INT_UP + 0.275LN_CAP_UP + 0.005LN_N_UP

Note: LOW, MID and UP, represent the lower, middle and upper Vaal catchment areas, LN_COST is the natural log of the total agricultural costs, LN_CAP is the natural log of capital costs, LN_INT is the natural log of intermediate costs and LN_N is the level of nitrogen present in the water.

Source: de Villiers (2009)

The model behaves as expected and all estimated coefficients were significant. A change in the intermediate cost (of 1%) leads to a change in the total cost (of 0.727%) in the same direction. This means that an increase in intermediate cost leads to an increase in total agricultural costs. The effect for capital cost is similar, except that a change in capital cost leads to smaller change in total agricultural cost: a 1% change in the capital cost results in a 0.275% change in total agricultural costs in the same direction.

Eutrophication plays a significant part in the model as a whole in explaining total cost, but the effect of eutrophication is relatively small. According to the model, a 1% increase in the nitrogen level of the raw water results in only a 0.005% increase in the total cost of production.

4.1.2 Industry cost model

As discussed in section 3.2.2, data collected for the industry cost model did not allow for econometric analysis. It was, however, established, through discussions with Eskom and Sasol, that water quality in general and not eutrophication *per se*, is the dominant concern for these industries as the treatment process for salt indirectly addresses the symptoms of eutrophication. To address this, it was decided to rather use data regarding the cost incurred by water supply services due to eutrophication and to transfer these costs back to mines and industries through the tariffs paid for the water consumed.

4.1.3 Water treatment cost model

4.1.3.1 Costs incurred by water supply services

The water treatment cost model results as estimated by de Villiers (2009) are given in Box 3.

Box 3: Ordinary least squares regression results for the water supply services sector (Rand Water), Vaal Dam, 2000-2008

LN_C_L = 1.567*** + 0.145**LN_N + 0.227**LN_P

Note: LN_C_L is the natural log of the cost per kilolitre to treat water, LN_P is the natural log of the phosphorus level present in water and LN_N is the natural log of the nitrogen level present in water. *** Denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%.

Source: de Villiers (2009)

According to this model, changes in the independent variables together explain 60% of the change in the dependent variable over time. The signs and magnitudes of the estimated coefficients make theoretical sense in terms of the literature reviewed and information gathered in the course of the study. From the model results, a 1% increase in nitrogen levels leads to a 0.145% increase in the costs to treat the water. Furthermore, a 1% increase in the level of phosphorus present in the water results in a 0.227% increase in the cost per kilolitre incurred by Rand Water.

Using the water treatment costs model (Box 3) and information regarding the operating costs of Rand Water (Table 4.1) it is possible to convert the % changes into monetary values.

Table 4.1: Rand Water operating costs, 2000-2008.

Year	Operating costs (R)	Volume treated (kL)	Operating costs (R/kL)
2000	855792.00	557846.90	1.53
2001	987562.00	588344.77	1.68
2002	1031534.00	649376.00	1.59
2003	1192036.00	709743.38	1.68
2004	1326278.00	743393.69	1.78
2005	1407972.00	712023.39	1.98
2006	1410699.00	718824.87	1.96
2007	1568090.00	768479.54	2.04
2008	1677703.00	759226.58	**2.21**

Source: de Villiers (2009)

From the model (Box 3), a 10% rise in the level of nitrogen present in the water will lead to a 1.45% increase in water treatment (operating) costs. Considering 2008 operating costs (R2.21 /kL), a 10% increase in nitrogen levels present in the water will result in a 3.2c increase in operating costs per kilolitre of water treated. Similarly, a 10% increase in phosphorus levels present in the water will result in a 5.0c/kL increase in operating costs. While these increases may appear negligible, when multiplied by the total volume of water treated they are, in fact, significant.

It is likely that any increase in treatment cost will be shifted directly to the end-consumer through the tariffs charged by water supply services for water supplied. End-consumers include municipalities (in the event that they do not treat water themselves), mines, industries and households.

4.1.3.2 The relationship between raw water quality and the chemical costs of producing potable water

The study predicted chemical water treatment costs at Zuikerbosch (station #2) and Balkfontein from the observed level of raw water quality variables. Descriptive statistics reveal that raw water in the Vaal River is of a poorer quality at Balkfontein compared to that at Zuikerbosch. Furthermore, the actual real chemical water treatment costs (measured in 2006 ZAR) averaged R0.09 /kL at Zuikerbosch and R0.13/kL at Balkfontein, indicating that the chemical water treatment costs of producing potable water tend to increase as raw water quality declines. The level of chlorophyll (total photosynthetic pigments) for Zuikerbosch ranged from 0.97 μg/L to 12.64 μg/L. On the other hand, Balkfontein water treatment station in the Middle Vaal Water Management Area experienced levels of chlorophyll ‘a’ from 5.76 μg/L to 267.80 μg/L, indicating that water eutrophication is a relatively smaller problem in the Upper Vaal Water Management Area compared to the Middle Vaal Water Management Area.

For both water treatment plants, ordinary least squares regression was used to estimate the relationship between real chemical costs of water treatment and the dimensions of water quality identified through principal component analysis of the water quality variables. The estimated regression models account for over 50.2% and 34.7% of variation in real chemical water treatment costs at Zuikerbosch and Balkfontein, respectively.

For predictive rather than explanatory purposes, a partial adjustment regression model was estimated for each of the two water treatment plants. Using this model, real chemical water treatment costs were specified as a function of real chemical water treatment costs in the previous time period and of raw water quality variables in the current period. The R^2 statistics for the two regression models were 61.4% using the data for Zuikerbosch and 59.9% using the data for Balkfontein, suggesting that both models have reasonable levels of predictive power. From the predictive models estimated for Balkfontein and Zuikerbosch, the chemical costs of water treatment for Zuikerbosch and Balkfontein are predicted at R0.096 /kL and R0.091 /kL respectively. The partial adjustment model is not suitable as an explanatory model of water chemical treatment costs and the model should not be used to predict water treatment costs for raw water qualities significantly different from those experienced in the Upper Vaal Water Management Area during the period November 2004-October 2006.

The chemical water treatment cost at Balkfontein is predicted to decrease by more than 3% for a 1% increase in the raw water loading of calcium, keeping all other variables constant. A 1% rise in water temperature, keeping all other variables constant, is predicted to increase chemical water treatment cost by 1.886%. At Balkfontein, chemical water treatment cost is expected to increase on average by 0.346% for a 1% increase in the level of chlorophyll 'a' and 2.077% for a 1% increase in turbidity. From the partial adjustment model it is apparent that calcium and turbidity are the main drivers of chemical costs of water treatment. An increase of 1% in raw water turbidity at Balkfontein could raise chemical water treatment cost by (R0.001895*360 000 kL*365 days) = R250 000 per annum (at average 2004-2006 costs), provided that Balkfontein treats water at its full capacity (i.e. 360 000 kL per day). In the same way, an increase in the level of calcium content in raw water by 1% could save Balkfontein R361 000 per annum (at average 2004-2006 costs).

At Zuikerbosch, chemical water treatment costs can be predicted from four water quality variables, namely nitrate, total alkalinity, electrical conductivity and sulphate, and previous period (week) cost. An increase of 1% in nitrate, keeping all other variables constant, is predicted to increase real water chemical treatment costs by less than 0.3%. An increase of 1% in total alkalinity loading in raw water is predicted to decrease real water chemical treatment cost by 0.223%, all other variable kept constant. The study shows that, if raw water nitrate in the Upper Vaal Water Management Area increases by 1%, keeping all other variables constant, chemical water treatment costs at Zuikerbosch are predicted to increase by (R0.00285*1 998 000 kL*365 days) = R208 000 per annum (at average 2004-2006 costs) – provided that Zuikerbosch treats water at a daily average of 1 998 000 kL per day. If Zuikerbosch operates at its daily average capacity and is able to keep the optimum level of total alkalinity in the Upper Vaal Water Management Area (thereby reducing the need for lime dosages to treat water), the estimated saving on chemical water treatment cost could be in the region of R156 000 per annum (at average 2004-2006 costs).

4.1.3.3 Application of the water treatment cost model

The general model specified by Gebremedhin (2009) to investigate the relationship between raw water quality and the chemical costs of producing potable water for the Zuikerbosch and Balkfontein water treatment plants (this report Section 4.1.3.2), has subsequently been used in another WRC study of the positive and negative consequences associated with the introduction of zero-phosphate detergents into South Africa (Quayle et al., 2010). The model was used to predict the base treatment costs for raw water abstracted from the respective dams (i.e. treatment costs in the absence of the application of zero-phosphate detergents). The model was re-estimated based on the application of zero-phosphate detergents to the respective systems to generate a second estimation of treatment costs. Assuming all other water quality parameters remained on average the same, this allowed an estimation of the likely reductions in water treatment costs associated with the introduction of zero-phosphate detergents (Quayle et al., 2010).

Based on the water treatment cost model and volumes of water treated, the current base level treatment costs and the reduction in water treatment costs (due to the application of zero-phosphate detergents) were estimated for the respective dams. The results are summarised in Table 4.2. The application of the water treatment cost model developed in this study in estimating water treatment costs and potential cost savings (as a result of adopting new practices) demonstrates how the research undertaken in this study can be used in a practical manner.

Table 4.2: Summary of water treatment costs (savings) for selected dams on the basis of the application of zero-phosphate detergents.

Dam	**Hartbeespoort**	**Roodeplaat**	**Klipvoor**	**Total**
Volume treated (ML/day)	90	90	5	
Predicted decline in Chlorophyll 'a' concentration (%)	22.2	23.3	21.3	
Base treatment cost (no phosphate reduction) (R)	108	90	134	
Reduced treatment cost (zero-phosphate detergents used) (R)	97	83	118	
Estimated annual cost saving (R)	350 516	236 641	28 978	**616 134**

Source: Quayle et al. (2010)

4.1.4 Property price model

The property price model results, as estimated by Mostert (2009), are given in Box 4. The estimated property price model (Box 4) has an R^2 and adjusted R^2 value of 75% and 62% respectively which means that 62% of the variance in the unit price (dependent variable) is explained by variance in the explanatory variables. Furthermore, the model has a significant F-statistic (p-value < 0.05). These statistics indicate a reasonably good model fit.

Box 4: Property price model, Vaal catchment area, 1996-2006

$$UnitP_B = 0.714^{*} + 0.870^{**}LN\_CHLA\_B - 1.066^{***}LN\_NH_4\_B + 0.124LN\_NO_2NO_3\_B$$

$$UnitP_V = 1.064^{*} + 0.097^{*}LN\_CHLA\_V - 0.260^{**}LN\_NH_4\_V + 0.450^{***}LN\_NO_2NO_3\_V$$

$$UnitP_G = 2.867^{*} + 0.375LN\_CHLA\_G - 0.455^{**}LN\_NH_4\_G - 0.030LN\_NO_2NO_3\_G$$

Note: UnitP is the natural log of the sales price of property, divided by the stand size, LN_ CHLA is the natural log of the level of chlorophyll 'a' present in the water, LN_ NH_4 is the natural log of the level of ammonia present in the water, LN_NO_2NO_3 is the natural log of the level of nitrate and nitrite present in the water and B, V and G indicate the Bloemhof, Vaal and Grootdraai Dams respectively. *** Denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%.

Source: Mostert (2009)

The estimated coefficient for the CHLA variable is statistically significant at the 5% level (p-value < 0.05) for the Bloemhof area and very close to being statistically significant at the 5% level (p-value = 0.0574) for the Vaal Dam area. It also carries a positive sign in all three equations indicating that there is a positive relationship between unit price and chlorophyll 'a' level which is surprising. For example, in the Bloemhof Dam area, an increase of 1% in the chlorophyll 'a' level present in the water would lead to a 0.87% increase in the unit price of property in the area.

The estimated coefficient for the NH_4 variable is statistically significant and carries a negative sign for all three areas, as expected. A 1% increase in the levels of ammonia present in the dam will subsequently reduce house prices by 1.07% in the Bloemhof area, by 0.26% in the Vaal area and 0.46% in the Grootdraai area. For the NO_2NO_3 variable, only the estimated coefficient for the Vaal Dam was statistically significant (p-value = 0.007). However, it carried a positive sign indicating that an increase of 1% in nitrate and nitrite levels will cause a 0.45% increase in house prices, which is unexpected.

Economic theory recognises that often the dependent variable (in this case, the property price) does not adjust instantaneously to changes in the explanatory variables. Rather, the dependent variable very often responds to such changes with a lapse of time, or 'lag'. To investigate this possibility, the model was re-estimated using the same explanatory variables, but with a one year time lag (t_{-1}). The results are shown in Box 5. The one year lag model exhibits an R^2 and adjusted R^2 value of 78% and 65% respectively which is a relatively good fit and an improvement on the previous model.

The estimated coefficient for the CHLA variable is statistically significant for only the Vaal area and again positive, indicating that a 1% increase in chlorophyll 'a' levels will result in an increase in the unit property price of 0.13%. The NH_4 variable in this model shows mixed results. The estimated coefficients for the NH_4 variable are statistically significant for the Bloemhof and Vaal areas. The estimated coefficient for the Bloemhof area is positive indicating that for an increase of 1% in the level of ammonia present in the water, the unit property price will rise by 1.39% which is unexpected. For the Vaal Dam area, an increase of 1% in ammonia present in the water leads to a decreases in unit prices of property of 0.32%. The estimated coefficient for the NO_2NO_3 variable was statistically significant only in the Vaal area model. However, it was positive indicating that a 1% increase in the level of nitrates and nitrites present in the Vaal Dam will cause an increase in unit property prices in the Vaal area of 0.58%. Once again, this result is unexpected.

Box 5: Property price model with a one year lag in the explanatory variables, Vaal catchment area, 1996-2006

$$UnitP_B = 10.088^{***} + 0.299LN\_CHLA\_B_{t-1} + 1.389^{***}LN\_NH_4\_B_{t-1} + 0.731LN\_NO_2NO_3\_B_{t-1}$$

$$UnitP_V = 4.285^{***} + 0.128^{**}LN\_CHLA\_V_{t-1} - 0.320^{***}LN\_NH_4\_V_{t-1} + 0.583^{***}LN\_NO_2NO_3\_V_{t-1}$$

$$UnitP_G = 5.289^{***} + 0.114LN\_CHLA\_G_{t-1} - 0.097LN\_NH_4\_G_{t-1} + 0.379LN\_NO_2NO_3\_G_{t-1}$$

Note: UnitP is the natural log of the sales price of property, divided by the stand size, LN_ CHLA is the natural log of the level of chlorophyll 'a' present in the water, LN_ NH_4 is the natural log of the level of ammonia present in the water, LN_NO_2NO_3 is the natural log of the level of nitrate and nitrite present in the water and B, V and G indicate the Bloemhof, Vaal and Grootdraai Dams respectively. *** Denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%.

Source: Mostert (2009)

4.1.5 Recreation value model

The average willingness to pay (WTP) for improved water quality for all three study sites was calculated by Mostert (2009) using the contingent valuation survey data and measured in Rand value per person, per night's stay at one of the study dams. Using a value-loss relationship model the average WTP values for the three study sites were estimated as R36 for the Bloemhof Dam, R53 for the Vaal Dam and R66 for the Grootdraai Dam, per person per nights stay (Mostert, 2009).

However, because of the relatively small sample size of the surveys, it was necessary to allow for an upper and lower bound, or confidence level. Allowing for a 30% confidence level and rounding to the nearest rand, the following upper and lower bounds for each of the study areas were estimated:

- R25 < Bloemhof < R47
- R37 < Vaal < R69
- R46 < Grootdraai < R86, (Mostert, 2009).

These estimates can be interpreted as the average value or range of values a visitor to the respective dam would be willing to pay to improve the water quality of the dam per night's stay. For example, a visitor to the Vaal Dam would be, on average, willing to pay R53 per person, per night's stay to improve the water quality of the Vaal Dam.

In response to survey questions regarding the water quality of the three dams, 68% of visitors to the Vaal Dam indicated that they were concerned about the water quality of the dam. Similarly, visitors to the Grootdraai Dam pointed to water quality as a problem with 45% of respondents showing concern regarding the quality of its water. However, none of the respondents to the Bloemhof Dam survey indicated concern for its water quality (Mostert, 2009). The Bloemhof Dam is one of the few dams in the Vaal River system that has flood gates; this may explain the lack of concern over water quality in this area as opening the flood gates washes away algae and foul smells. A more detailed discussion of the survey responses is available in Mostert (2009).

4.2 Integrated model results

The relationship between eutrophication and the various sectors (property, agriculture, water treatment and recreation) affected by eutrophication was difficult to establish. This was mainly due to the quality of the data used in this study and the availability of data for a longer period (however, this is addressed under Section 4.2.2). The expected relationships are outlined in Table 4.3.

A negative relationship between two variables implies that a change in the one will cause a change in the other in the opposite direction, whereas a positive relationship indicates that both variables will change in the same direction. A negative relationship is expected between eutrophication and property prices: an *increase* in eutrophication is expected to result in a *decrease* in property prices. A positive relationship is expected between eutrophication and water treatment costs: an *increase* in eutrophication is expected to result in an *increase* in water treatment costs. Table 4.2 shows that eutrophication is mainly expected to present a cost to the property, agricultural and water treatment sectors.

Table 4.3: Postulated relationship between eutrophication and the impacted economic sector.

Impacted sector	Eutrophication variables	Expected relationship	Explanation
Property sector (property prices)	Chl-a NH_4 PO_4	Negative relationship Negative relationship Negative relationship	Eutrophication is expected to have a negative impact on surrounding property prices.
Agricultural sector (agricultural costs)	NH_4 PO_4	Positive or negative relationship Positive relationship	The expected impact of NH_4 can be positive or negative since NH_4 is a natural fertiliser. However, similar to PO_4 it can increase irrigation costs of farmers.
Water treatment sector (water treatment costs)	NH_4 PO_4	Positive relationship Positive relationship	NH_4 and PO_4 are expected to increase costs of the water treatment sector.
Recreational sector (recreational activities)	Chl-a NH_4 PO_4	Negative relationship Negative relationship Negative relationship	Eutrophication is expected to have a negative impact on surrounding recreational activities.

Note: Chl-a is the level of chlorophyll 'a' present in the water; NH_4 the concentration of ammonia present in the water; and PO_4 is the concentration of phosphorus (as phosphate) present in the water.

Recreational sector cost data were not available over a longer period of time. These data were collected primarily at a point in time (see Mostert, 2009) and were not available for the entire study period. Therefore, the integrated model analysis does not include the recreational sector due to data limitations as stated. Additionally, as mentioned previously, it is difficult to understand the full impact of the data due to the averaging effect of econometric models (given the cyclical nature of eutrophication data).

4.2.1 Model results, 1996-2006

Seemingly unrelated regression (SUR) models were estimated for each of the dams (Box 6 to 8) for the period 1996 to 2006. This period was chosen to establish a baseline for eutrophication costs in recent times. The SUR models were estimated in log linear form, therefore, the coefficients can be interpreted directly as elasticities, that is, a 1% change in an explanatory variable in the model leads to an X% change in the specific cost associated with that particular equation. The results reveal that the nature of the relationship between eutrophication and impacted sector costs was as expected (Table 4.4) for some dams, but not for all.

In particular, the property equations were limited in adhering to both the expected relationships and the significance of estimated coefficients. This is not surprising given the myriad of factors that affect property prices. However, some useful features did emerge from this analysis with the most important being the clear negative impact of Chl-a on property prices (**Chl-a** is the **main driver of water discoloration** and is a product of eutrophication).The results also reveal the nature of the impact of PO_4 and NH_4 on the agricultural and water treatment sectors. As expected NH_4 acts either as a cost or as a benefit to the agricultural sector, but mostly as a benefit across all the dams. Furthermore, PO_4 acts as a cost to the agricultural sector. The water treatment sector is largely impacted by **PO_4 – the main driver of eutrophication**.

The Bloemhof Dam results (Box 6) do not reveal a good fit, which is not unexpected given the number of factors that drive the various economic costs. Statistically, the results are also poor with the majority of the variables being statistically insignificant. The model exhibits serial correlation which influences the statistical significance of the explanatory variables negatively. Serial correlation occurs when the error terms from different (usually adjacent) time periods are correlated. Serial correlation arises in time-series studies when the errors associated with a given time period carry over into future time periods.

Box 6: Integrated model results, cost relationship estimates for the Bloemhof Dam, 1996-2006

Total costs to property = 13.45** - 1.04**Chl-a$ – 0.09$NH_4$$ – 1.17$PO_4$$

Total agriculture cost = 5.85** + 0.07NH_4 + 0.36$PO_4$$

Total water treatment costs = -0.41 + 0.08 NH_4 + 0.15* PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the first lag.

As was the case with the Bloemhof Dam, the results for the Grootdraai Dam (Box 7) are poor. This can be attributed to the relatively small sample size. As indicated in Section 4.2.2, a larger sample size does provide better results. The Grootdraai Dam results were negatively influenced by serial correlation which necessitated a functional transformation of the property price equation (Sibande, 2010). This improved the overall significance of the variables and the statistical significance of the models. A dummy variable was introduced into the property equation for the Grootdraai Dam model to account for the structural break (level change) in the dependent variable (property price): there was a rapid increase in property price from 1996 to 1998 (i.e. a structural break in the data) and moderation from then until 2006. Hence, the dependent variable is not 'strictly' linear and without the inclusion of the dummy variable, the relationship could be estimated as statistically insignificant.

It must be noted that SUR models have a stringent requirement namely that the random errors of each equation in the models must have constant variance and no serial correlation. The models performed reasonably well against this requirement. However, some of them exhibited a weak positive serial correlation with the exception of the Grootdraai property sector equation which was transformed to account for a strong positive serial correlation.

Box 7: Integrated model results, cost relationship estimates for the Grootdraai Dam, 1996-2006

Total costs to property = 5.79** + 0.0032Chl-a + 1.45***Dummy$ – 0.0014NH_4 – 0.0035$PO_4$$

Total agriculture cost = 5.82** - 0.36***NH_4 + 0.99***$PO_4$$

Total water treatment costs = -0.47 + 0.21***NH_4 + 0.025PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the first lag.

An interesting feature of the models is the difference in the timing of the impact on the costs of the different sectors. For example, in the Bloemhof Dam the impact of eutrophication on the agricultural sector is realised at the second lag (i.e. total costs to agriculture now are a function of eutrophication levels two years earlier), while for the Grootdraai Dam the impact on costs is more immediate at the first lag (i.e. total costs are now a function of eutrophication levels one year earlier).

Among the three equations, the Vaal Dam model (Box 8) performed the best. However, the agricultural equation is not a strong one. The models, similar to the two other study sites, suffer from weak to strong serial correlation. The results, also similar to the other study sites, reveal a sensitivity to sample size (see also Section 4.2.2).

Box 8: Integrated model results, cost relationship estimates for the Vaal Dam, 1996-2006

Total costs to property = -1.21 - 0.63***Chl-a$ + 0.99$NH_4$$ + 1.12$PO_4$$
Total agriculture cost = 7.71*** - 0.35**NH_4 + 0.32$PO_4$$
Total water treatment costs = 0.18 - 0.15***NH_4 + 0.12***PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the first lag. & means that the figures were used in the second lag.

4.2.2 Model extension to include 2007 and 2008

The estimation period was extended from 1996-2006 to 1996-2008. The purpose of this extension was to establish the consistency of the relationships in the baseline models and to determine whether the estimation period of the baseline models had a negative bias on the results. The extended models were estimated in exactly the same manner as the baseline models. This was done to allow for direct comparison of the baseline and extended models. The additional data were obtained from the same sources as the initial data and were, therefore, subject to similar challenges to the initial data. The results from the model extension are presented in Boxes 9 to11.

A comparison of the extended models to the baseline models shows that the fit and the significance of the results improved in some of the cases, but not all. This can be seen clearly in the Bloemhof Dam agricultural equation, for example, where the R^2 improved and the estimated coefficient for PO_4 is now statistically significant. The extension of the study period, however, did not improve the statistical significance of either the Grootdraai Dam property sector equation or the Vaal Dam water treatment costs equation.

The extended models, however, did confirm that the nature of the relationship between eutrophication and the costs of impacted economic sectors is complicated. Firstly, the results vary from dam to dam suggesting that either the quality of the data differs from dam to dam or eutrophication cannot be explicitly captured to the same extent in each of the dams. Regardless of these challenges, the extended models highlight to some degree the impact that eutrophication has on the various sectors of the economy.

Box 9: Extension of the integrated model, cost relationship estimates for the Bloemhof Dam, 1996-2008

Total costs to property = 13.03*** - 1.02***Chl-a$ - 0.24$NH_4$$ - 0.91$PO_4$$
Total agriculture cost = 5.32*** + 0.17NH_4 + 0.39**$PO_4$$
Total water treatment costs = -0.57 + 0.09**NH_4 + 0.17*PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the second lag.

Box 10: Extension of the integrated model, cost relationship estimates for the Grootdraai Dam, 1996-2008

Total costs to property = 5.52*** + 0.0002Chl-a$ + 1.47***Dummy + 0.015$NH_4$$ + 0.054$PO_4$$
Total agriculture cost = 3.59*** - 0.26***NH_4 + 1.51***$PO_4$$
Total water treatment costs = -0.24 + 0.18***NH_4 + 0.0014***PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the first lag.

Box 11: Extension of the integrated model, cost relationship estimates for the Vaal Dam, 1996-2008

Total costs to property = 2.44 - 0.71**Chl-a + 1.06$NH_4$$ + 0.09$PO_4$$
Total agriculture cost = 6.02*** - 0.08NH_4 + 0.50**$PO_4$$
Total water treatment costs = -0.38 + 0.24*NH_4 - 0.016PO_4

Note: *** denotes significance at 1%; ** denotes significance at 5%; * denotes significance at 10%. Figures all in logarithm. $ means that the figures were used in the first lag. $ means that the figures were used in the second lag.

4.2.3 Economic costs associated with eutrophication

Applying the extended models, the Rand value equivalent of the economic costs associated with eutrophication was calculated for each dam and are summarised in Tables 4.5 to 4.7. Essentially this calculation is the actual cost of the average level of eutrophication in a given dam. The calculations of the Rand costs are limited by the level of significance of the extended models, therefore, some of the calculated costs are only indicative as they relate to statistically insignificant variable coefficients. The variables with statistically significant coefficients are highlighted in the tables. The period of analysis was from 1997 to 2008.

The average costs to agriculture per year, as a result of phosphorus present in the water (average phosphorus levels over the study period), were estimated at R247/ha for the Bloemhof Dam, R455/ha for the Grootdraai Dam and R145/ha for the Vaal Dam. Grootdraai agriculture has a higher sensitivity (or elasticity) to changes in the level of phosphorus present in the water than agriculture in the other dam areas.

The property sector was most affected by chlorophyll ‘a’ levels (annual average chlorophyll ‘a’ levels over the study period) the average cost to the property sector was estimated as R854/m^2 in the Bloemhof Dam area and R3340/m^2 in the Vaal Dam area. These costs are calculated directly from the property price data and changes in property prices. For all the dams, property prices increased significantly over the observed period and, therefore, the costs (benefits) were calculated in line with these increases. For Grootdraai, the models in all scenarios indicated a negligible (and statistically insignificant) parameter estimate, meaning that the rand value of a 1% change in ‘Chl-a’ is minimal for property in the Grootdraai Dam area.

The average costs (over the study period) to the water treatment sector per dam, as a result of ammonia levels, were estimated to be R0.18/kL for the Bloemhof Dam, R 0.0015/kL for the Grootdraai Dam and R0.0011/kL for the Vaal Dam. Table 4.4 presents the descriptive statistics of the eutrophication data used in this analysis and shows the range of compound levels (chlorophyll ‘a’, ammonia and phosphorus – as phosphate) over which the above costs are estimated.

However, it is difficult to comprehend the full impact of eutrophication on these sectors from the models developed in this study as the daily eutrophication data were converted to yearly averages effectively smoothing out the variability and peaks within the eutrophication data. As a result of averaging the eutrophication data, the coefficients for the eutrophication variables were likely to have been underestimated and the model results should be interpreted in the context of this limitation.

Table 4.4: Descriptive statistics of the levels of the compounds used as a proxy of eutrophication, 1996-2008 in the integrated model.

Compound levels (mg/L)	Minimum	Maximum	Average
		Bloemhof Dam	
Chl-a	7.32	65.82	23.30
NH_4	31.82	398.13	118.13
PO_4	24.77	157.45	52.03
		Grootdraai Dam	
Chl-a	3.37	19.67	10.35
NH_4	22.13	418.71	101.99
PO_4	27.84	80.14	42.40
		Vaal Dam	
Chl-a	0.67	22.00	8.33
NH_4	24.40	117.48	55.37
PO_4	25.55	100.00	50.33

Table 4.5: Integrated model application, total cost to the property sector per dam (Rand per m^2), 1997-2008.

Year	Bloemhof Dam			Grootdraai Dam			Vaal Dam		
	Chl-a	NH_4	PO_4	Chl-a	NH_4	PO_4	Chl-a	NH_4	PO_4
	Statistically significant at 1% level;	Not statistically significant;	Not statistically significant;	Not statistically significant;	Not statistically significant;	Not statistically significant;	Statistically significant at 5% level;	Not statistically significant;	Not statistically significant;
	Sign according to expectation	Sign according to expectation	Sign according to expectation	Sign not according to expectation	Sign not according to expectation	Sign not according to expectation	Sign according to expectation	Sign not according to expectation	Sign not according to expectation
1997	-854.10	-467.26	-498.64	0.02	2.19	1.53	-324.60	78.17	4.21
1998	-518.04	-287.11	-302.44	0.02	2.44	1.70	-388.44	93.54	5.03
1999	-458.10	-253.90	-267.45	0.09	10.42	7.28	-316.24	76.16	4.10
2000	-212.26	-117.64	-123.92	0.09	10.40	7.27	-219.86	52.95	2.85
2001	-236.54	-131.10	-138.09	0.09	10.44	7.30	-194.11	46.75	2.51
2002	-346.62	-192.11	-202.36	0.09	10.53	7.36	-37202.8	8 959.09	481.97
2003	-183.31	-28.28	-107.02	0.09	10.46	7.30	-733.49	176.64	9.50
2004	-120.68	-66.89	-70.46	0.09	10.40	7.27	-145.25	34.98	1.88
2005	-7 833.39	-4 341.51	- 4 573.24	0.09	10.37	7.25	-122.18	29.42	1.58
2006	-95.66	-53.02	-55.85	0.09	10.38	7.25	-140.72	33.89	1.82
2007	-76.74	-42.53	-44.80	0.09	10.41	7.27	-135.28	32.58	1.75
2008	-79.20	-43.90	-46.24	0.10	11.65	8.14	-151.46	36.47	1.96
Average	-854.10	-467.26	-4.99	0.08	9.17	6.41	- 3 339.54	804.22	43.26

Note: a negative sign (-) implies a reduction in property prices.

Table 4.6: Integrated model application, total agricultural cost per dam (Rands per hectare), 1997-2008.

Year	Bloemhof Dam		Grootdraai Dam		Vaal Dam	
	NH_4	PO_4	NH_4	PO_4	NH_4	PO_4
	Not statistically significant; Sign according to expectation	Statistically significant at 5% level of significance; Sign according to expectation	Statistically significant at 1% level of significance; Sign according to expectation	Statistically significant at 1% level of significance; Sign according to expectation	Not statistically significant; Sign according to expectation	Statistically significant at 5% level of significance; Sign according to expectation
1997	359.30	236.91	-515.80	576.44	-47.87	142.86
1998	398.74	262.92	-503.80	563.03	-53.13	158.55
1999	297.31	196.03	-408.05	456.03	-39.61	118.21
2000	367.29	242.18	-546.22	610.44	-48.94	146.04
2001	257.43	169.74	-303.61	339.31	-34.30	102.36
2002	188.25	124.12	-217.96	243.59	-23.77	70.92
2003	362.08	238.74	-396.11	442.68	-43.37	129.43
2004	515.21	339.71	-478.21	534.43	-64.92	193.74
2005	593.40	391.27	-548.99	613.53	-76.59	228.55
2006	322.99	212.97	-302.90	338.51	-42.36	126.40
2007	293.99	193.85	-249.43	278.76	-35.58	106.18
2008	544.85	359.25	-410.02	458.22	-71.77	214.18
Average	375.07	247.31	-406.76	454.58	-48.52	144.78

Note: a negative sign (-) refers to a benefit provided by that particular variable.

Table 4.7: Integrated model application, total water treatment cost per dam, (Rands per kilo litre), 1997-2008.

Year	Bloemhof Dam		Grootdraai Dam		Vaal Dam	
	NH_4	PO_4	NH_4	PO_4	NH_4	PO_4
	Statistically significant at 5%; Sign according to expectation	Statistically significant at 5%; Sign according to expectation	Statistically significant at 1%; Sign according to expectation	Not statistically significant; Sign according to expectation	Statistically significant at 5%; Sign according to expectation	Not statistically significant; Sign not according to expectation
1997	0.07	0.14	0.00109	0.00216	0.00082	-0.002622
1998	0.07	0.15	0.00116	0.00229	0.00087	-0.002786
1999	0.08	0.15	0.00123	0.00243	0.00092	-0.002949
2000	0.08	0.16	0.00130	0.00256	0.00097	-0.003113
2001	0.09	0.17	0.00137	0.00270	0.00102	-0.003277
2002	0.09	0.19	0.00150	0.00295	0.00112	-0.003586
2003	0.09	0.18	0.00141	0.00279	0.00106	-0.003393
2004	0.09	0.19	0.00150	0.00295	0.00112	-0.003588
2005	0.10	0.20	0.00159	0.00314	0.00119	-0.003811
2006	0.11	0.22	0.00176	0.00348	0.00132	-0.004224
2007	0.11	0.22	0.00175	0.00345	0.00131	-0.004192
2008	0.11	0.23	0.00182	0.00359	0.00136	-0.004359
Average	0.18	0.092	0.0015	0.00023	0.00109	-0.0035

Note: a negative sign (-) refers to a benefit provided by that particular variable.

4.2.4 Cost of exceeding the Resource Water Quality Objectives

The second application of the expanded models was to determine the costs of exceeding the Department of Water Affairs' Resource Water Quality Objectives (RWQO). The RWQO were set for phosphates and chlorophyll 'a' as indicated in Tables 4.8 to 4.10. These objectives were determined under the Department of Water Affairs "Development of an Integrated Water Quality Management Plan for the Vaal River System" programme (Directorate National Water Resource Planning, 2009). The RWQOs indicate the limit of the assimilative capacity of the dams that are within the Vaal River system (Bloemhof, Grootdraai and Vaal Dams included).

The level at which the actual levels of phosphates and chlorophyll 'a' exceed the RWQOs represents the costs of eutrophication. Highlighted rows indicate a year in which the RWQOs have been exceeded. This section focuses on the costs of exceeding the RWQOs. The calculation of the costs is based on the Rand value cost of a 1% increase of phosphates and chlorophyll 'a' (as set out in the models above). The costs to society were calculated on a per hectare basis for the agricultural sector, per meter squared basis for the property sector and per kilo litre basis for the water treatment sector.

The results reveal that both phosphate and chlorophyll 'a' levels regularly exceed the RWQOs. From the Bloemhof Dam results (Table 4.8) it is evident that chlorophyll 'a' levels exceed the RWQOs for four of the 12 study years and phosphorus RWQO levels are exceeded for seven of the 12 study years. In the Grootdraai Dam (Table 4.9) phosphorus levels exceed the RWQOs for three of the 12 study years. The results from the Vaal Dam analysis (Table 4.10) indicate that the RWQOs for chlorophyll 'a' and phosphorus were exceeded for two and four of the 12 study years, respectively. The costs depend on the extent to which the RWQOs are exceeded. Here it is important to note the limitation mentioned previously, namely that annual average eutrophication statistics have been used. The costs of exceeding the RWQOs are, therefore, also linked to only those years where the average RWQO has been exceeded for an entire year. Costs, however, are linked to the variability and seasonality of rapid and short-term increases (spikes) in concentration levels of these various eutrophication compounds. Irrespective of these shortcomings, it was possible to determine functional relationships between eutrophication and economic costs, especially linked to years in which the average eutrophication level exceeded the stated RWQO. The costs varied from relatively low to as much as R2 900 /ha/year for agriculture, R1.44 /kL for water treatment and R18 800 /m^2 with respect to residential property prices.

Table 4.8: Cost of exceeding RWQOs for the Bloemhof Dam, 1997-2008.

	Bloemhof Dam				
	Actual value	RWQO (μg/L)	Percentage difference	Rand value for a 1% change (R)	Rand value cost (R)
	Chl-a (Property – cost per metre squared)				
1997	7.33	30	-76%	10.43	
1998	8.12	30	-73%	9.22	
1999	32.43	30	8%	4.27	34.60
2000	12.02	30	-60%	4.76	
2001	7.32	30	-76%	6.98	
2002	19.98	30	-33%	3.69	
2003	61.90	30	106%	2.43	258.26
2004	65.82	30	119%	157.65	18 821.32
2005	15.01	30	-50%	1.93	
2006	13.31	30	-56%	1.54	
2007	37.09	30	24%	1.59	37.67
2008	12.72	30	-58%	1.78	
	PO_4 (Agricultural – cost per hectare)				
1997	25.88	30	-14%	7.27	
1998	26.39	30	-12%	8.07	
1999	26.36	30	-12%	6.01	
2000	44.41	30	48%	7.43	356.93
2001	29.25	30	-2%	5.21	
2002	34.33	30	14%	3.81	54.92
2003	24.77	30	-17%	7.32	
2004	37.52	30	25%	10.42	261.38
2005	32.43	30	8%	12.00	97.18
2006	157.45	30	425%	6.53	2 775.73
2007	82.80	30	176%	5.95	1 046.78
2008	109.50	30	265%	11.02	2 920.76

Table 4.8: *continued*

	Actual value	RWQO (μg/L)	Percentage difference	Rand value for a 1% change (R)	Rand value cost (R)
			PO_4 (Water treatment – cost per kilo litre)		
1997	25.88	30	-14%	0.0021	
1998	26.39	30	-12%	0.0022	
1999	26.36	30	-12%	0.0024	
2000	44.41	30	48%	0.0025	0.12
2001	29.25	30	-2%	0.0026	
2002	34.33	30	14%	0.0029	
2003	24.77	30	-17%	0.0027	
2004	37.52	30	25%	0.0029	0.07
2005	32.43	30	8%	0.0031	0.02
2006	157.45	30	425%	0.0034	1.44
2007	82.80	30	176%	0.0034	0.59
2008	109.50	30	265%	0.0035	0.93

Table 4.9: Cost of exceeding RWQOs for the Grootdraai Dam, 1997-2008.

	Grootdraai Dam				
	Actual value	RWQO (μg/L)	Percentage difference	Rand value for a 1% change (R)	Rand value cost (R)
			PO_4 (Agriculture – cost per hectare)		
1997	29.06	50	-42%	59.49	
1998	36.44	50	-27%	58.11	
1999	27.84	50	-44%	47.06	
2000	36.94	50	-26%	63.00	
2001	40.00	50	-20%	35.02	
2002	45.00	50	-10%	25.14	
2003	51.10	50	2%	45.69	100.51
2004	31.00	50	-38%	55.16	
2005	36.95	50	-26%	63.32	
2006	43.55	50	-13%	34.94	
2007	55.89	50	12%	28.77	338.65
2008	80.14	50	60%	47.29	2 850.90
			PO_4 (Water treatment – cost per kilo litre)		
1997	29.06	50	-42%	0.00002	
1998	36.44	50	-27%	0.00002	
1999	27.84	50	-44%	0.00002	
2000	36.94	50	-26%	0.00002	
2001	40.00	50	-20%	0.00002	
2002	45.00	50	-10%	0.00002	
2003	51.10	50	2%	0.00002	0.000051
2004	31.00	50	-38%	0.00002	
2005	36.95	50	-26%	0.00003	
2006	43.55	50	-13%	0.00003	
2007	55.89	50	12%	0.00003	0.00034
2008	80.14	50	60%	0.00003	0.00180

Table 4.10: Cost of exceeding RWQOs for the Vaal Dam, 1997-2008.

	Vaal Dam				
	Actual value	RWQO (μg/L)	Percentage difference	Rand value for a 1% change (R)	Rand value cost (R)
			Chl-a (Property – cost per metre squared)		
1997	22.00	20	10%	1.96	19.63
1998	3.25	20	-84%	2.35	
1999	4.98	20	-75%	1.91	
2000	0.67	20	-97%	1.33	
2001	8.92	20	-55%	1.17	
2002	6.93	20	-65%	224.96	
2003	12.36	20	-38%	4.44	
2004	18.35	20	-8%	0.88	
2005	3.54	20	-82%	0.74	
2006	2.62	20	-87%	0.85	
2007	20.39	20	2%	0.82	1.58
2008	3.10	20	-85%	0.92	
			PO_4 (Agriculture – cost per hectare)		
1997	36.84	50	-26%	10.84	
1998	27.72	50	-45%	12.03	
1999	25.55	50	-49%	8.97	
2000	100.00	50	100%	11.08	1 108.02
2001	64.21	50	28%	7.77	220.68
2002	43.53	50	-13%	5.38	
2003	42.59	50	-15%	9.82	
2004	47.15	50	-6%	14.70	
2005	55.49	50	11%	17.34	190.24
2006	91.26	50	83%	9.59	791.32
2007	40.13	50	-20%	8.06	
2008	25.86	50	-48%	16.25	

5. DISCUSSION OF THE COSTS ASSOCIATED WITH EUTROPHICATION

The objective of this project was to construct a generic model that could be used to assess the costs associated with eutrophication and to apply it to the Vaal River system. During the development of the various economic sector models it became clear that the econometric approach adopted for this purpose was unable to capture the full costs of eutrophication to each of the sectors considered. The main reason for this limitation is most likely linked to the data issues discussed in Section 1.4, specifically the averaging of water quality observations to annual averages which fails to capture the variability in eutrophication and the consequences of this variability to the various economic sectors. As a result, the costs of eutrophication estimated from the models developed in this study only capture a portion of the total costs of eutrophication and are, therefore, an underestimation of the total economic impact of eutrophication. The aim of this chapter of the study is to put the costs of eutrophication estimated in this study into perspective in terms of total economic costs of eutrophication and to develop an understanding of the full economic impact of eutrophication.

5.1 Water treatment sector

In this section, two examples of the impact of severe eutrophication on water treatment costs are presented. These costs, along with the cost of eutrophication to the water treatment sector estimated using regression analysis (Section 4.1.3), are illustrated graphically in Figure 5.2 to show the relative impact of different levels of eutrophication on water treatment costs.

A case study from Rand Water estimated a Geosmin incident in the Vaal River catchment in September 1995, to cost Rand Water in the region of R5 million (Swanepoel & Du Preez, 2006). Geosmin is an organic compound produced by several classes of microbes, including cyanobacteria. The costs included R240 000 for increased sample analysis, R3.6 million for the loss of already-treated water, R500 000 for discarded backwash water and R500 000 for increased pumping costs due to changes in loading from Zuikerbosch to Vereeniging. In Figure 5.1, increased levels of Geosmin and Chlorophyll (total photosynthetic pigments) are evident during September and October of 2005 and again in November 2005. This case study clearly illustrates the significant impact of a once-off eutrophication event on water treatment costs.

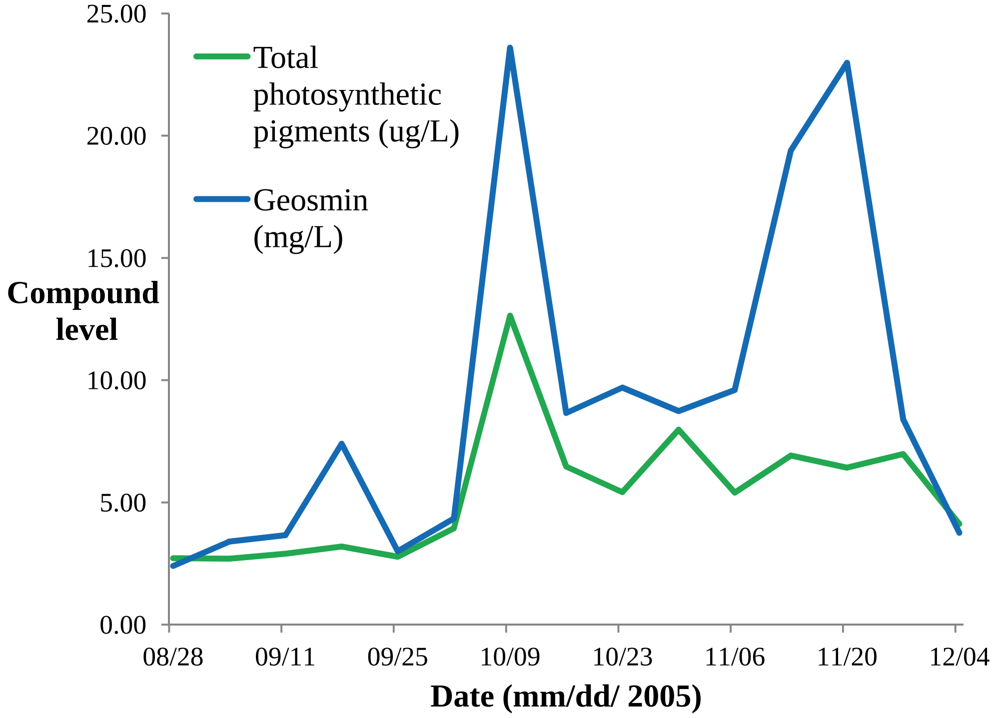

Figure 5.1: Graph of weekly Geosmin and Chlorophyll (total photosynthetic pigment) levels at the Zuikerbosch Water Treatment Plant, September to December, 2005.

In another study, an investigation of the impact of algal blooms on water treatment costs, costs were determined based on actual data obtained in laboratory and pilot scale tests that were carried out on a water source that was low in organic contaminants (total organic carbon content of between 4 and 8 mg/L) as well as on the same water after it had been spiked with cyanobacterial algal scums to contain algal cell concentrations of between 10 000 and 500 000 cells/mL. Tests were also conducted on a water source that was polluted largely by industrial effluents and containing a fairly high total organic carbon concentration (15 to 35 mg/L). The laboratory tests included both enhanced coagulation and advanced treatment processes (Freese et al., 2001) while pilot plant testing investigated advanced treatment options. Table 5.1 summarises the results of the study for a range of volumes of raw water treated.

Calculating the increases in water treatment costs that arise from algal blooms can only be done for actual case studies, since the factors affecting treatment costs are both numerous and varied, as well as being localised. An attempt has been made here to quantify this increase, although in this case too, the results give only an indication for an actual case study. The results obtained for this study indicate that for advanced treatment options, the cost per ML water treated decreases as the size of the water treatment works increases, which is to be expected since the economy of scale effect becomes important for large plants.

The results also indicate that enhanced coagulation is a more cost-effective option. However, it should be borne in mind that although enhanced coagulation was found to be as effective as the advanced treatment technologies in terms of organic contaminant removal measured as TOC, it was not as effective in removing micropollutants such as taste and odour compounds (e.g. geosmin and 2-methyliso borneol) or for algal toxins. It may therefore be necessary to use the more expensive option of advanced treatment when severe algal blooms occur. Perhaps even more important is the fact that this study indicates that cyanobacterial blooms can be expected to increase treatment costs from R0.08 /kL to R0.43 /kL (enhanced coagulation treatment) and even to R0.69 /kL (advanced treatment). Effectively, this is a five to nine times increase in costs, respectively, or in percentage terms an approximate 400% to 750% increase, respectively, for the duration of the cyanobacterial bloom. Prevention of the occurrence of algal blooms has very significant water treatment cost benefits, which need to be included in any cost estimate measures to reduce nutrient enrichment of susceptible water resources.

Table 5.1: Laboratory investigation of the impact of a cyanobacterial bloom on water treatment costs: the costs of treating non-eutrophic and eutrophic water.

Treatment volumes (kL/day)	Costs of treating non eutrophic water (R/kL)	Costs of enhanced coagulation treatment of eutrophic water (R/kL)	Costs of advanced treatment of eutrophic water (R/kL)
10 000	0.0802	0.4278	0.6904
25 000	0.0802	0.4278	0.678
50 000	0.0802	0.4278	0.6683
100 000	0.0802	0.4278	0.6582
500 000	0.0802	0.4278	0.6334
1 000 000	0.0802	0.4278	0.6221

Table 5.2 brings together all the water treatment cost studies: the model results obtained in this study and the two examples discussed in this section. For each study, the results or cost effects are summarised as percentage changes in costs.

Table 5.2: A summary of the estimated relationships between water treatment costs and eutrophication.

Study	**Result**
Costs incurred by water supply services (Section 4.1.3.1)	1% ↑ in N → 0.14% ↑ in operating costs (Rand Water) 1% ↑ in P → 0.226% ↑ in operating costs (Rand Water)
Raw water quality and chemical costs of producing potable water (Section 4.1.3.2)	1% ↑ in N → 0.3 ↑ in water treatment costs (Zuikerbosch Treatment Plant) 1% ↑ in Chl ’a’ → 0.346% ↑ water treatment costs (Balkfontein Treatment Plant)
Rand Water Geosmin incident (Section 5.1)	↑ Geosmin levels → 72% ↑ in Rand Water’s costs
The impact of algal blooms on water treatment costs (Section 5.1)	Cyanobacterial bloom → 400% (enhanced coagulation treatment) to 740% (of advanced treatment) ↑ in treatment costs

These relationships are displayed graphically, Figure 5.2, to illustrate how the various estimates of costs fit together to form a more complete assessment of the impact of eutrophication on water treatment costs. The first two studies (Sections 4.1.3.1 and 4.1.3.2) consider changes in water treatment costs as a result of relatively small changes in eutrophication (1% changes). These changes would fall into the first ‘band’ of eutrophication levels as shown in Figure 5.2. The Rand Water Geosmin incident (Section 5.1) shows a significant increase in water treatment costs, with a ‘jump’ in costs of approximately 72% (band 2), based on 2005 Rand Water operating costs (Table 4.1) The laboratory investigation of the impacts of cyanobacterial blooms on water treatment costs (Section 5.1, Table 5.1) describes an additional significant (short-term) jump in water treatment costs of 400 to 750% (3 and 4 in Figure 5.2) depending on the type of water treatment technique required.

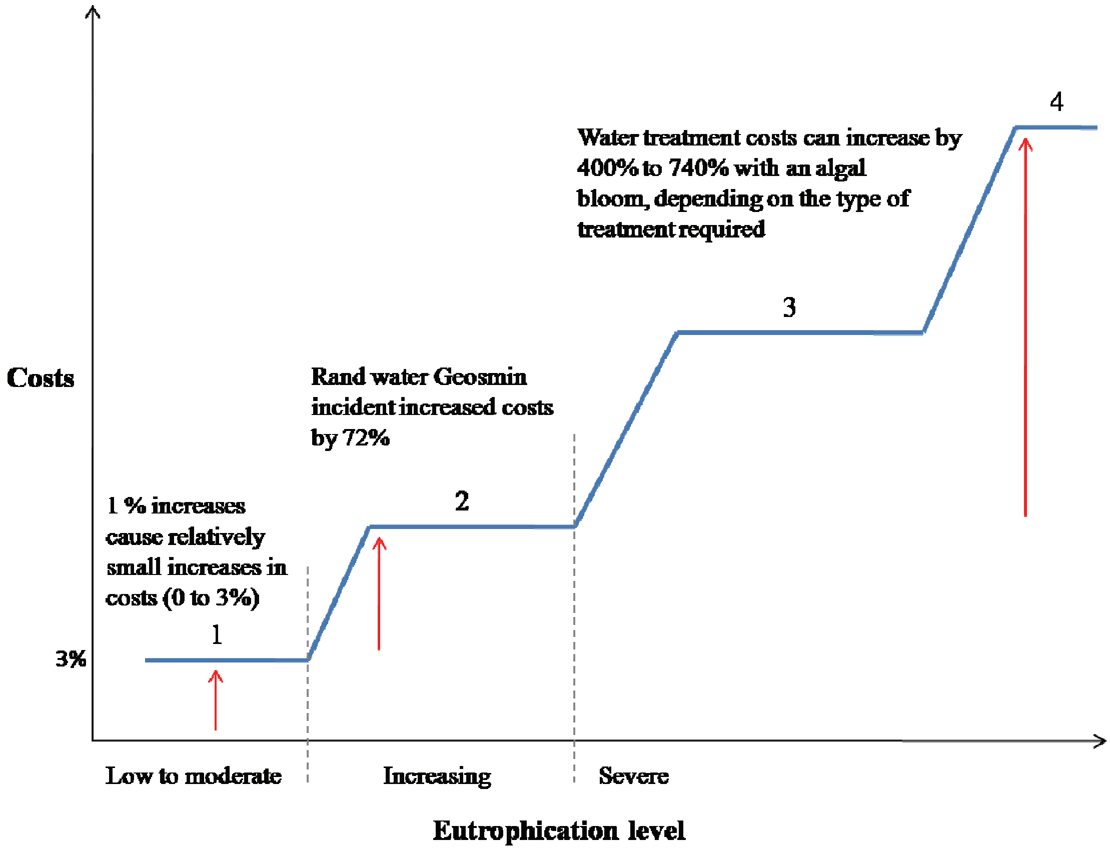

Figure 5.2: A theoretical depiction of the relationship between water treatment costs and levels of eutrophication.

The 'large jumps' in water treatment costs at high levels of eutrophication can be explained, in part, by considering the chemical cost composition of treating raw water (as shown by the significant increases in costs associated with enhanced and advanced water treatment processes). For example, in a study of water quality in impoundments within the Umgeni Water operational area, Graham et al. (1998) described the average chemical cost composition for Durban Heights water treatment works from 1990 to 1997 and for a particularly severe period of taste and odour formation in March 1994 (Figure 5.3). During the severe period of taste and odour formation there is a substantial increase in the use of activated carbon to treat the water. The price of activated carbon is significantly higher than that of the other water treatment chemicals, therefore, an increase in the amount of activated carbon used in the treatment process would add significantly to the water treatment cost.

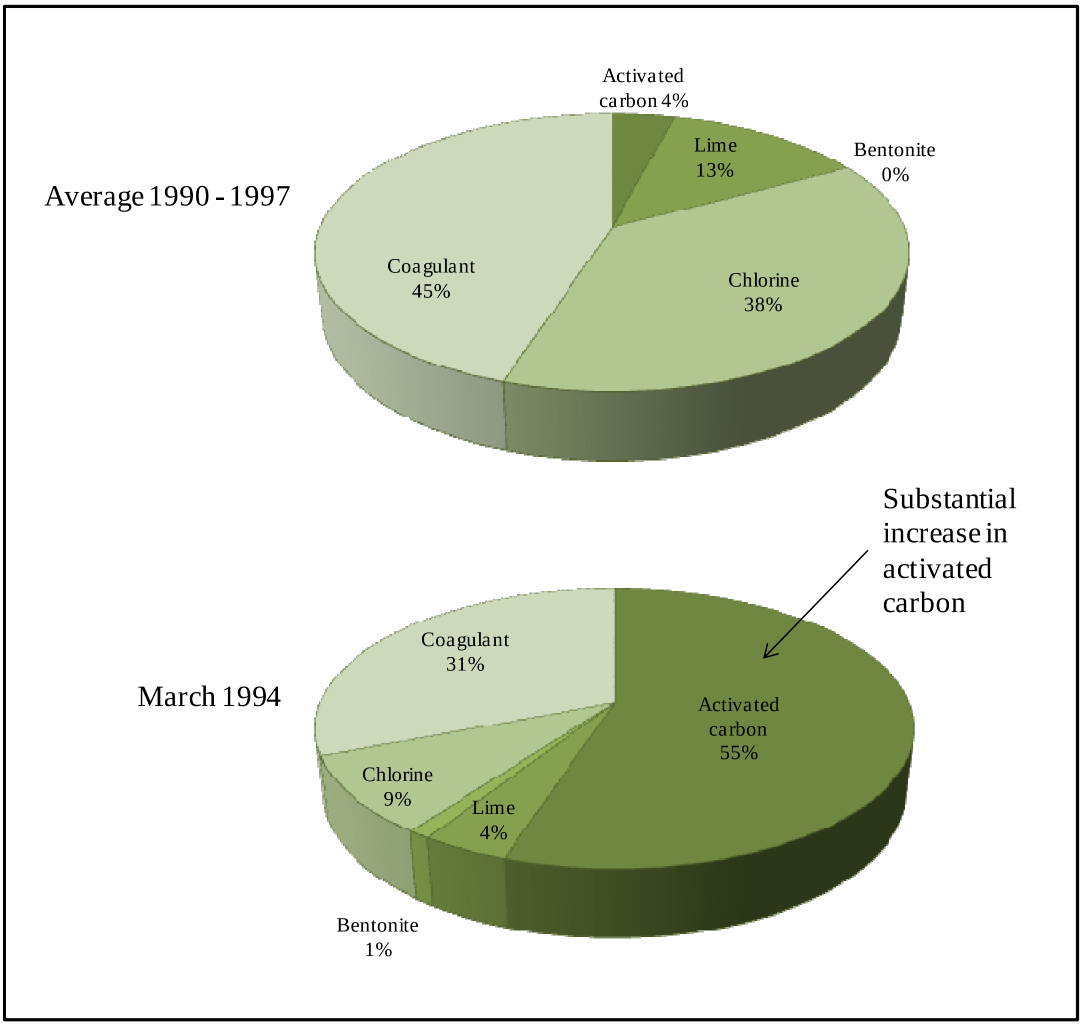

Figure 5.3: Water treatment cost compositions for Durban Heights Water treatment works on average for 1990-1997 and for March 1994 during a particularly severe period of taste and odour formation.

Source: Graham et al. (1998)

This section, particularly Figure 5.2, brings together the costs to the water treatment sector of various levels of eutrophication and shows that the cost estimates by the models developed within this study only capture a portion of the total impact of eutrophication to the water treatment sector. The econometric models confirm a definitive cost impact on the water treatment sector due to eutrophication, even though the methodology applied is somewhat limited in capturing the full costs of eutrophication. The additional examples discussed in this section (5.1) indicate that the full impact of eutrophication on the water treatment sector is greater than that captured by the models developed and applied in this study.

Additionally, a cost, not shown in Figure 5.2, is the capital cost of infrastructure required to deal with future eutrophication events. This capital investment wouldn't be necessary in the absence of eutrophication. This cost is a long term liability to the water treatment sector and is difficult to establish, however, it would further increase the total costs of eutrophication to the water treatment sector.

5.2 Agricultural sector

In the agricultural cost model (Sections 3.2.1 and 4.1.1), only the levels of phosphorus and nitrogen (classical eutrophication precursors) present in the water were used to approximate eutrophication. The model results show that nitrogen levels are statistically significant in explaining total costs to agriculture, but that the effect is relatively small. However, annual average nitrogen levels are used, effectively smoothing out the impacts of once-off, but severe, incidents of eutrophication, such as algal blooms which may contaminate water bodies with potentially toxic compounds. The costs of such events to the agricultural sector are not fully captured in the econometric models and may significantly increase the total costs of eutrophication to the agricultural sector. A few examples from the literature are presented below to illustrate the significance of eutrophication to the agricultural sector.

There are a number of references to livestock deaths related to the ingestion of water contaminated with cyanobacteria. In South Africa, Harding and Paxton (2001) detail the death of seven stock and domestic animals due to cyanobacterial toxicosis between December 1993 and December 1996 in the south Western Cape. During the same period a single stock loss was reported in the Northern Province. The financial losses as a result of these animal deaths were estimated to be R1.1 million (Harding and Paxton, 2001). Further incidents linked to cyanobacteria contamination include: the loss of 290 in-milk dairy cows in the Kareedouw district and the slaughter of a further 70 animals diagnosed with acute photosensitivity; the deaths of three lambs in the Malmesbury district; and the loss of 18 goats in the Northern Province (Harding and Paxton, 2001). In Australia, Steffensen et al. (1999) documented thirteen cases of stock deaths related to cyanobacterial blooms and in one of these, 1600 livestock deaths were reported (Department of Water Resources, 1992). The use of water contaminated by algal blooms for irrigation purposes is also a concern: the extent of the impact of toxins in water on crops and on humans and livestock who consume these crops is unclear.

Figure 5.4 depicts the theoretical relationship between agricultural costs and levels of eutrophication. The figure indicates the level of eutrophication and its associated costs captured by the econometric models applied in this study, as well as the potential costs to the agricultural sector that are not captured by the models – the costs linked to short-term severe incidents of eutrophication, such as algal blooms.

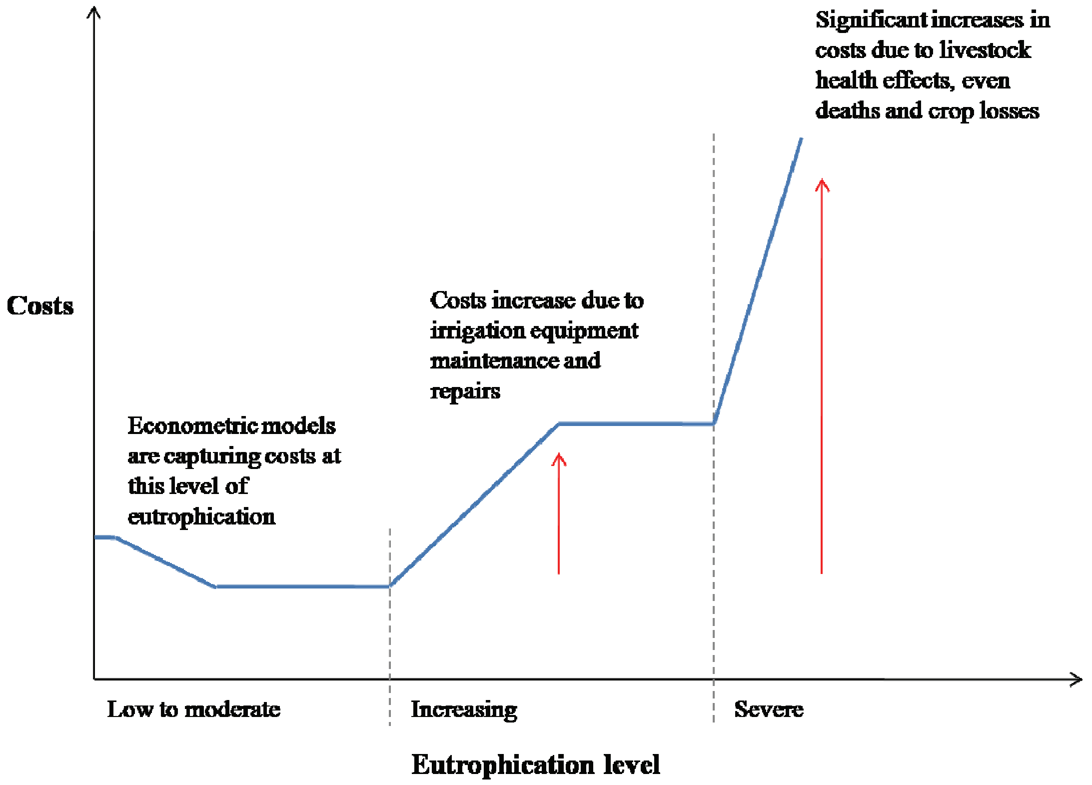

Figure 5.4: A theoretical depiction of the relationship between agricultural costs and levels of eutrophication.

At low to moderate levels of eutrophication, costs to agriculture are not significantly increased, instead raised levels of phosphorus in irrigation water can promote crop growth and decrease costs through reduced fertiliser requirements. This is illustrated in Figure 5.4 as a decrease in costs at low to moderate eutrophication levels. However, after a certain level of eutrophication these benefits are likely to be outweighed by increasing costs linked to repeated irrigation equipment repairs or replacements (Smajstria, 1995) and eventually to detrimental health impacts on livestock and crops (Dennison and Lyne, 1997).

5.3 Property sector

The property sector model was again developed using average annual eutrophication data and considered the levels of ammonia, nitrate and nitrite present in the water as an indication of eutrophication. The model estimated a statistically significant, but weak relationship between property value and levels of ammonia present in the water. However, many eutrophication indicators (such as nitrogen levels) are not easily observed (by property owners) and do not impair the utility derived from waterside properties. Thus, property owners are often unaware of eutrophication changes until high nutrient concentrations combine with other factors to cause algal blooms, which reduce water clarity and which can cause algal scums on the surface of the water and taste and odour issues (Legget and Bockstael, 2000). Thus, the response of the property market to eutrophication is likely to occur only at severe levels of eutrophication. This concept is illustrated in Figure 5.5.

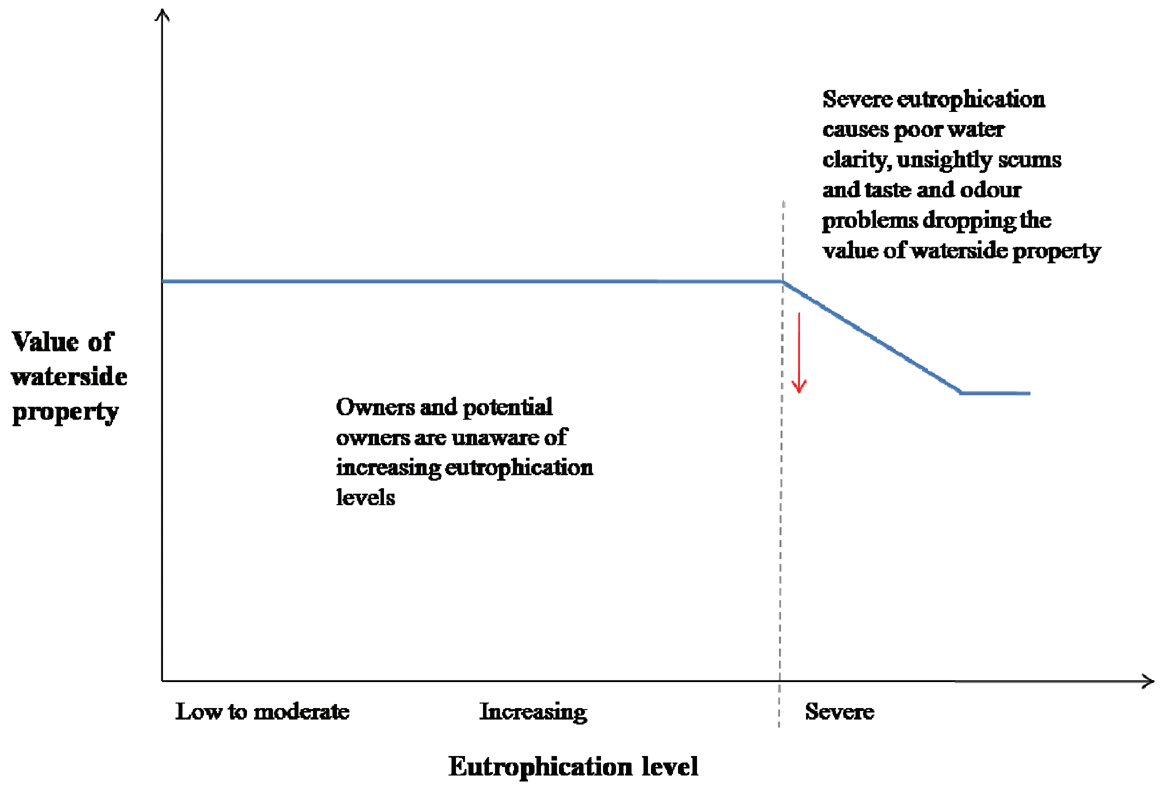

Figure 5.5: A theoretical depiction of the relationship between property value and levels of eutrophication.

The impact of water quality and especially eutrophication on property prices in South Africa has not yet been fully investigated. A study by Van der Walt et al. (2010) considered the determinants of house prices in Hout Bay, South Africa. In the study, the proximity of the property to the Hout Bay River and the presence of odours were two factors expected to negatively influence house prices, although the effect of water quality was not investigated directly. However, a number of international studies have shown poor water quality to negatively influence property value (see Chapter 2 and DNRE, 2000; Legget and Bockstael, 2000; Poor et al., 2001; Pretty et al., 2002). For example, a study of the price of water front properties around Lake Boga in Australia found that property values dropped by 20 to 25% after major algal blooms in the Lake (DNRE, 2000).

5.4 Recreation

In this study (Section 4.1.5), the average willingness to pay for improved water quality for all three study sites was calculated and the results showed that visitors to each of the dams were willing to pay between R25 and R86 pppn for improved water quality. However, the study did not consider the impact of severe eutrophication on the tourism industry in the event that either of the dams (or all) became completely unusable for recreation purposes. In the case of severe eutrophication, if a water body was closed to recreationists due to the presence of potentially toxic compounds, the entire recreation value would be lost. This could impact severely on local tourism and even have knock-on effects at the national level. As Pretty et al. (2002) explain, value loss costs are incurred, not only when people are prevented by eutrophication and algal blooms from enjoying the quality of the water body, but also by those people whose livelihoods rely on visitors who would otherwise have used the clean water body. Consequently, when eutrophication prevents access, this revenue is lost.

Studies quantifying the relationship between declines in tourism in South Africa and eutrophication are, at present, relatively uncommon. Several studies, quantifying the impact of eutrophication on the tourism sector, can be found for Australia (Hudnell, 2008):

- The Darling River, 1991, where *Anabaena* species covered 1000 km of the Darling River in central Australia, where losses to the tourist industry were estimated to be around Australian $1.5 million.
- The Hawksbury Nepean River near Sydney in New South Wales where estimates indicated losses of around Australian $6.7 million in 1991/1992 compared to the previous summer when no blooms had occurred. It is interesting to note that the blooms were not classified as toxic, but that the revenue losses were largely due to negative publicity surrounding the algal bloom problem.
- In 1991, nine storage sites within New South Wales which were used for recreational purposes experienced algal blooms that were estimated to have resulted in revenue losses of around Australian $1.2 million.

Figure 5.6 is an illustration of the relationship between the recreation value of a water body and its level of eutrophication. At low to moderate levels of eutrophication, the impact on recreation of eutrophication may be relatively minimal as recreationists are unaware of rising eutrophication levels.

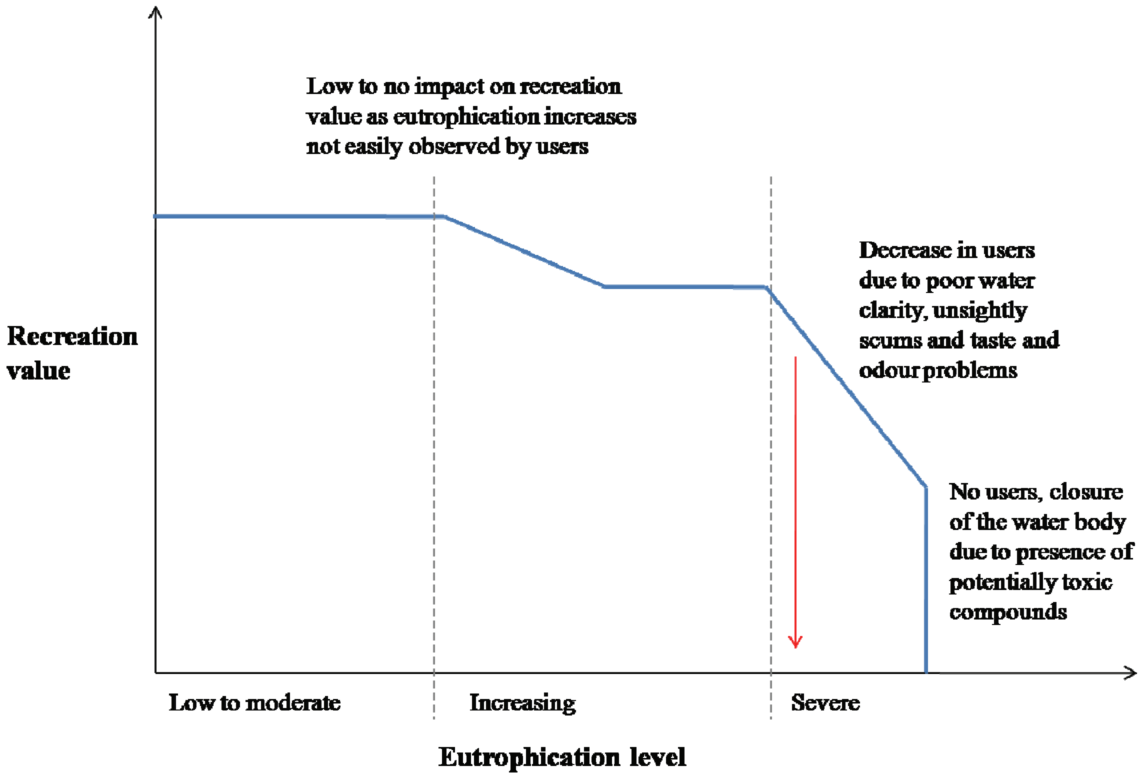

Figure 5.6: A theoretical depiction of the relationship between recreation value and levels of eutrophication.

However, at severe eutrophication levels when algal blooms are likely to occur, the recreation value of the water body may drop rapidly as users are discouraged by poor water clarity, unsightly scums and taste and odour problems. In the event of an algal bloom the water body may be officially closed to recreationists and other users owing to the presence of potentially toxic compounds in the water. In this case, the water body would lose its recreation value affecting recreational users and all those whose livelihoods rely on visitors who would otherwise have used the clean water body.

6. CONCLUSIONS AND RECOMMENDATIONS

The objective of this project was to develop a generic model to assess the costs associated with eutrophication in South Africa and to apply it to the Vaal River system. From the various studies undertaken within this project it is clear that the relationship between water quality and economic indicators is complex. Changes in water quality depend on a system of interdependent climatological, anthropogenic, physical and biological factors and the feedback of the culmination of these factors to impacts (and hence changes) in the economy is a slow and complex process. This feedback involves the interaction between two completely different systems, one physical by nature and the other economic. Each additional variable adds another layer of complexity and increases the information requirement if this feedback is to be fully understood.

Given this complexity it turned out to be relatively difficult to assign an economic cost to eutrophication as economic costs and impacts are usually linked to coarse, high-level, annual impacts whereas eutrophication events are site-specific and very time-specific occurrences. A case study from Rand Water (1995) estimated that an incident in the Vaal River catchment of increased Geosmin levels in the water cost Rand Water in the region of R5 million (Swanepoel & Du Preez, 2006). Geosmin is an organic compound produced by several algae, including cyanobacteria. Economic variables, such as house prices and the cost of agriculture, are also impacted by a host of both global and local events, as well as long-term trends. To isolate the link between such long-term economic trends and specific events of eutrophication using average annual data necessitates that the relationships found are weak. This suggests that while there might be site-specific and event-specific costs, the economy-wide effects are difficult to ascertain in this manner.

In terms of the association between agricultural costs and eutrophication, the estimated relationship was relatively weak; however, it was shown that eutrophication did contribute to total agricultural costs. Furthermore, it is likely that this relationship was actually underestimated as a result of data and model limitations (as discussed in Section 1.4). It is expected that as eutrophication levels rise, the relationship between agricultural costs and eutrophication would strengthen and the impact of eutrophication on the agricultural sector would be greater. Given that the output of irrigated agriculture in the Vaal system contributes significantly to country-wide food production and that water is a primary input in irrigation farming, increased costs of eutrophication would not only affect the agricultural sector, but would have knock-on effects for the economy.

From the potable water supply services model, it is clear that eutrophication contributes to total water treatment costs. The estimated relationship between total costs and eutrophication (nitrogen and phosphorus levels) was statistically significant and showed that a 10% increase in nitrogen present in the water would result in an increase in water treatment costs of 3.2c/kL. A similar increase in phosphorus present in the water would lead to a 5.0c/kL increase in water treatment costs. It is likely that any increase in water treatment costs will be shifted directly to the end-consumer through the tariffs charged by water supply services for water supplied.

For the Balkfontein water treatment plant, the study identified total hardness, calcium content and turbidity as the main drivers of chemical cost of water treatment. An increase of 1% in raw water turbidity at Balkfontein could raise chemical water treatment costs by (R0.001895*360 000 kL*365 days) R250 000 per annum (at average 2004-2006 costs), provided that Balkfontein treats water at its full capacity (i.e. 360 000 kL per day). In the same way, an increase in the level of calcium content in raw water by 1% could save Balkfontein R361 000 per annum (at average 2004-2006 costs).

For the Zuikerbosch water treatment plant, the study predicted that an increase in raw water nitrate in the Upper Vaal Water Management Area of 1%, keeping all other variables constant, would cause chemical water treatment costs at Zuikerbosch to increase by (R0.00285*1 998 000 kL*365 days) R208 000 per annum (at average 2004-2006 costs) – provided that Zuikerbosch treats water at a daily average of 1 998 000 kL per day. If Zuikerbosch operates at its daily average capacity and is able to keep the optimum level of total alkalinity in the Upper Vaal Water Management Area (thereby reducing the need for lime dosages to treat water), the estimated saving on chemical water treatment cost could be in the region of R156 000 per annum (at average 2004-2006 costs).

The property price models showed that increased levels of ammonia present in the water would lead to a reduction in the price of property for all three study areas. The estimated coefficients for the chlorophyll ‘a’ and nitrite/nitrate variables were generally not statistically significant. Property prices are influenced by eutrophication levels, but once again the strength of the estimated relationship was relatively weak. However, changes in property price, as a result of eutrophication, are most likely related to the worst case of water quality over an extended period and not to average water quality. It remains to ascertain the impact of eutrophication on property prices associated with occurrences of severe eutrophication.

Using conjoint analysis, willingness to pay (WTP) values for improvements in water quality were estimated for all three dams to ascertain the costs to recreation associated with eutrophication. It was found that visitors to the Bloemhof Dam would be willing to pay between R25 and R47 (per person per night’s stay) for water quality improvements, visitors to the Vaal Dam would be willing to pay between R37 and R69 and the WTP values for Grootdraai Dam were between R46 and R86. Furthermore, many of the respondents to the Vaal (68%) and Grootdraai (45%) Dam surveys indicated concern for the water quality of these two dams.

It was intended to investigate the ecosystem costs of eutrophication within this study and this objective fell under the initial PhD project. However, as a result of the restructuring of the study, these costs were not analysed due to a lack in capacity. Hence, the ecosystem costs of eutrophication still require investigation. Similarly, costs of eutrophication associated with eutrophication control and monitoring (laboratory) costs as well as costs linked to eutrophication and water quality policy and strategy development remain unclear and require exploration.

In terms of the integrated model, the relationship between eutrophication and the various sectors (property, agriculture and water treatment) affected by eutrophication was difficult to establish. The results reveal that the agricultural sector is most affected by eutrophication as it is directly dependent on water quality for irrigation. Property prices and the cost of water treatment are only marginally affected by eutrophication. However, it is difficult to comprehend the full impact of eutrophication on these sectors from the models developed in this study as the daily eutrophication data were converted to yearly averages effectively smoothing out the variability and peaks within the eutrophication data. Irrespective of these shortcomings, it was possible to determine functional relationships between eutrophication and economic costs, especially linked to years in which the average eutrophication level exceeded the stated Resource Water Quality Objectives (RWQO). The estimated costs vary from very small to as much as R2 900 /ha/year for agriculture, R1.44 /kL for water treatment and R18 800 /m^2 with respect to residential property prices.

In a separate investigation (Quayle et al., 2010), the water treatment cost model developed within this study (Section 4.1.3.2) was used to predict the chemical treatment costs for raw water abstracted from the Hartbeespoort, Roodeplaat and Klipvoor Dams. The model was then re-applied to generate a second estimation of treatment costs based on the application of zero-phosphate detergents to the respective systems. From the results, a combined total predicted saving of R616 000 per year was estimated for treating water from the Hartbeespoort, Roodeplaat and Klipvoor Dams. The application of the water treatment cost model developed in this study in estimating water treatment costs and potential cost savings (as a result of adopting new practices) demonstrates that the research undertaken in this study can be practically applied and used to inform policy and strategy development.

The various analyses within this study show that eutrophication has an economic impact on the sectors of agriculture, property, recreation and water treatment. However, several of these analyses underestimated the relationship between eutrophication and the particular economic sector. A discussion of incidents of severe eutrophication (Chapter 5), both locally and internationally, clearly illustrates the potentially significant costs of eutrophication to the various economic sectors. The models developed in this study do not capture the full impact of eutrophication on the economy. However, it can be expected that the costs of eutrophication presented in this study are even higher in reality. Furthermore, ecosystem costs of eutrophication and costs associated with the control and monitoring of eutrophication as well as costs linked to policy development around eutrophication, are yet to be determined.

It is, therefore, recommended that in addition to this study, two different kinds of analysis be undertaken. The one is the development of a systems-dynamic model in which both physical and economic variables are present and the linkages shown. The other is to develop a monitoring programme whereby eutrophication outbreaks are monitored, together with its plausible causes and the conditions that lead to it, and linked to event-specific costs. Eutrophication outbreaks are event driven, following spikes in nutrient loads. Economic data, however, by and large deals with longer term variables, monthly on an annual basis. To improve the mapping of the impact of eutrophication it is important to monitor outbreaks properly, seek to identify its cause and effects per event and to develop an “eutrophication memory" that can be used in the medium to longer term. Such an "eutrophication memory" is likely to provide invaluable information that would improve the understanding of eutrophication considerably.

In future studies of a similar nature, efforts must be made to resolve the data limitations related to eutrophication measurement variables. A possible method of overcoming these data issues is to apply ordinary spherical Kriging to the data as a spatial interpolation method (see Ara et al., 2006 for an application of this method), so as not to exclude the variability and peaks within eutrophication data from the analysis. Another option would be to model the relationship between economic sector costs as a result of severe eutrophication rather than average eutrophication levels – as it is often the worst case water quality that will drive the relationship (Dickens pers comm. 2011) – to understand the potential economic impacts of permitting eutrophication levels to rise.

Furthermore, studies of the ecosystem costs of eutrophication and the costs associated with the control and monitoring of eutrophication as well as costs linked to eutrophication policy and strategy development should be undertaken.

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8. APPENDICES

Appendix A: Eutrophication in the Vaal River system

	System	Level of problem	Type of problem	Principal cause	Action implemented
Upper Vaal	Vaal river at Parys	Persistently moderate	Algae and hyacinth	Sewage discharges and defuse pollution from agriculture	Special phosphate level of 1 mg/L
	Vaal river at Potchefstroom	Persistently moderate	Algae and hyacinth	Sewage discharges and defuse pollution from agriculture	Special phosphate level of 1 mg/L
	Klein Blesbokspruit	Low problem	Algae	Sewage discharge	Special phosphate level of 1 mg/L
	Blesbokspruit	Low problem	Weeds	Agricultural practice	Special phosphate level of 1 mg/L
	Vaal river at inflow from Bophelong	Persistently severe	Algae, hyacinth	Storm water from Bophelong	
	Upper Klip river	Persistently severe	Algae, weeds	Sewage discharge	Special phosphate level of 1 mg/L
	Rietspruit	Persistently severe	Algae, weeds	Sewage discharge	Special phosphate level of 1 mg/L
	Bocksburg lake	Persistently severe	Algae, water chemistry	Sewage discharge	Sample for phosphate
	Vaal river at Villiers	Infrequently severe	Algae, water chemistry	Sewage discharge	Special phosphate level of 1 mg/L
	Vaal Dam	Infrequently severe(low to moderate)	Blue green algae	Agricultural and sewage discharge	Report of spillage to DWAF, Assist with management of waste treatment works
Middle Vaal	Vaal river Barrage	Persistent severe (moderate to high)	All algae and macrophytes	Sewage discharge and informal settlements, high intensity farming	Physical removal of invader plants; Baley straw; reporting of spillage to DWAF; Assist with the management of wastewater treatment works
	Golf courses and urban impoundments	Persistent severe (moderately to high)	Algae and macrophytes	Sewage discharge and informal settlements, general run-off, contaminated ground water	

	Rivers including Klip River, Blesbokspruit	Persistent severe (moderately to high)	Macrophytes	Sewage discharges and informal settlements; high intensity farming	
Lower Vaal	Harts River	Seasonal moderate	Algae and water chemistry	Sewage discharges, surface run-off, irrigation backflows	
	Spitskop Dam	Moderate to extreme	Algae and water chemistry	Sewage discharges, surface run-off, irrigation backflows	Not yet in place as the dam is not extensively used in the in the area
	Lower Vaal River	Low to moderately	Algae, water chemistry	Upstream pollution	

Source: Walmsley (2005)

Appendix B: Commonly used indicators of eutrophication

Indicator	Explanation
Biological	
Phytoplankton algae	These are algae suspended, floating or moving only slightly under their own power in the water column. Often the dominant algae form in standing waters.
Chlorophyll 'a' concentration (µg/L)	Major light gathering pigment of all photosynthetic organisms imparting the characteristic colour or green plants. Its relative measurement in natural waters is indicative of the concentration of algae in the water.
Algal bloom frequency	Excessively large standing crop of algae, usually visible to the naked eye. More frequent blooms are indicative of nutrient enrichment.
Phytoplankton biomass	Biomass of algae suspended, floating or moving only slightly under their own power in the water column.
Periphyton algae	Attached forms of algae on a substrate for example, filamentous algae attached to rocks in streams.
Chlorophyll 'a' concentration (µg/mg)	The green pigment in periphyton algae, indicative of the concentration of algae attached to substrates in the water
Macrophytes	Higher plant, macroscopic plant, plant of higher taxonomic position than algae, usually a vascular plant. Aquatic macrophytes are those macrophytes that live completely or partially in water.
Presence of invasive aquatic weeds	Large stands of invasive aquatic weeds such as water hyacinth can be indicative of nutrient enrichment.
Chemical	
Nutrients	Substance that supports growth and reproduction. In the context of aquatic plants, these include nitrogen, phosphorus, carbon, silica and iron (among others).
Phosphorus – ortho-phosphate, Total-P	A macronutrient which appears in water bodies in combined forms known as ortho- and poly-phosphates and organic phosphorus. Phosphorus may enter a water body in agricultural runoff where fertilisers are used. Storm water runoff from highly urbanised areas, septic system leachate and lake bottom sediments also contribute phosphorus. A critical plant nutrient which is often targeted for control in eutrophication prevention plans.
Nitrogen – Nitrate/nitrite/ammonia	A macronutrient which occurs in the forms of organic nitrogen, ammonia nitrogen, nitrite nitrogen and nitrate nitrogen. Form of nitrogen is related to a successive decomposition reaction, each dependent on the preceding one, and the progress and decomposition can be determined in terms of the relative amounts of these four forms of nitrogen.

Indicator	Explanation
Cyanotoxin concentrations	Toxic substances (including cyclic peptides, alkaloids and lipopolysaccharides) produced by cyanobacteria as secondary metabolites.
Physical	
Water transparency	Transparency is a measure of how clear the water is. It is important because aquatic plants need sunlight for photosynthesis. The clearer the water, the deeper sunlight will penetrate. The photic zone depth is related to transparency. It is the illuminated zone, surface to depth beyond which light no longer penetrates. Generally equated with the zone in which photosynthetic algae can survive and grow, due to adequate light supply.
Turbidity	A measure of the light-scattering ability of water. It indicates the concentration of suspended solids in the water.
Suspended solids	Particles which can be removed by passing the water through a filter. Inorganic or organic matter, such as clay, minerals, decay products and living organisms, that remains in suspension in water. In surface waters it is usually associated with erosion or runoff after rainfall events. Suspended solids loadings are generally high in stream systems which are actively eroding a watershed. Excessive storm water runoff often results in high suspended solids loads to lakes. Many other pollutants such as phosphorus are often associated with suspended solids loadings. Phytoplankton represents organic suspended solids.
Dissolved oxygen	Refers to the uncombined oxygen in water which is available to aquatic life. Temperature affects the amount of oxygen which water can contain. Biological activity also controls the oxygen level. D.O. levels are generally highest during the afternoon and lowest just before sunrise.
Indices	
Trophic state	Refers to the degree of nutrient enrichment. The trophic status in impoundments is associated with total phosphorus and chlorophyll concentrations. The terms oligotrophic, mesotrophic, eutrophic, and hypetrophic are used to describe the trophic status of an impoundment.

Indicator	Explanation
Trophic state index	Trophic Status refers to the degree of nutrient enrichment and of the associated eutrophication problems of an aquatic ecosystem. The quantities of nitrogen, phosphorus, and other biologically useful nutrients are the primary determinants of a lake's trophic state index (TSI). Carlson's index is one of the more commonly used trophic indices. Because they tend to correlate, three independent variables can be used to calculate the Carlson Index: chlorophyll pigments, total phosphorus and Secchi depth.
Hypolimnetic oxygen demand	This is a measure of the oxygen demand exerted by decomposing organic matter, such as dead algae, at the bottom of deep, stratified reservoirs. Very productive reservoirs can have a high HOD.

Source: (DWAF, 2002, Mattson et al., 2004)