PRIORITIZATION OF RIVER BASINS IN THE TSHWANE AREA WITH REFERENCE TO FAECAL COLIFORM BACTERIA FOR THE PURPOSE OF THE IDENTIFICATION OF CANDIDATE WETLANDS FOR REHABILITATION

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ABSTRACT

Wetlands are considered a last line of defence against poor water quality. Despite the natural capabilities of wetlands to remove a variety of contaminants from surface water, the track record for wetland conservation leaves much to be desired. In the northern parts of the City of Tshwane, 84% of wetlands have been degraded. When viewed against the poor bacteriological quality of river water in the study area, the lack of wetland conservation efforts is of particular concern.

Given the large number of wetlands in the Tshwane area in need of rehabilitation, this study aimed to devise a methodology to prioritise these wetlands for rehabilitation. No blueprint for such a prioritisation process exists, as studies are adapted to take into account the availability of data and the unique requirements of the study area. The methodology for this study is based on the prioritisation of a specific river basin, based on expected maximum faecal bacterial load originating from various sources of pollution.

Four river basins were compared with each other in a series of screening processes. Screening was done on a landscape level using a Geographic Information System (GIS) to generate various composite layers as part of the screening process. The screening processes relied on the application of several weighted criteria. Weights for criteria are based on scientific literature. Weights are also allocated in line with the "worst case scenario", as the study is in essence an assessment of the various pollution sources and their maximum possible contribution to deteriorating surface water quality. A Simple Additive Weighting technique was used to assess the total pollution loads and total numbers of users at risk from contaminated surface water in each of the river basins. It is important to note that the objective is to only rate the pollution sources, whilst exact pollution loads were not calculated. Diffuse, areal and point sources of pollution were rated using the estimated contributions to faecal coliform loads. The river basin with the highest score

was selected for the selection of candidate wetlands for rehabilitation purposes.

The Apies River Basin scored highest for most of the criteria, with the exception of the number of households at risk from contaminated surface water. Despite the 0.60 weight allocated to households at risk, the extent of pollution sources in this river basin allowed it to be singled out as the basin in which a wetland for rehabilitation is most urgent in order to attenuate bacterial load. Two wetlands were short listed, based on their high need for rehabilitation, their hydrogeomorphic location (valley bottom with a channel), and given that they are larger than 1ha in size and within a minimum distance from the households at risk. Site level assessments are required for a final selection between the two, taking into account the nature of the current disturbances, the possibility of risk due to back-flooding, the projected costs associated with rehabilitation, the nature of the vegetation associated with the wetlands and the general conservation value of each of the wetlands.

DECLARATION

I declare that this research report, apart from the contributions mentioned in the acknowledgements, is my own, unaided work. It is being submitted for the Degree of Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other university.

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(Signature of candidate)

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CHAPTER 1

INTRODUCTION

1.1 OVERVIEW OF IMPORTANT CONCEPTS

A variety of terms are used to refer to various wetland types, including pan, swamp, estuary, mangrove, mire, marsh, vlei, bog, fen etc. Wetlands have been defined by Zedler and Kercher (2005) as areas where water is the primary factor controlling the environment and the associated plant and animal life. In contrast, the Ramsar Convention defined wetlands as ”areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters” (Ramsar Convention Secretariat, 2004). For the purposes of regulatory processes in the United States, wetlands must show evidence of all of the following: wetland hydrology, wetland soil and wetland plants (Zedler and Kercher, 2005). In South Africa, wetlands are protected under the National Water Act (South Africa, 1998, 10), which defines a wetland as: ”land which is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface, or the land is periodically covered with shallow water or would support vegetation typically adapted to life in saturated soils.” In contrast with the United States, South African wetlands only need to meet one of the criteria outlined by Zedler and Kercher (2005).

Wetlands have been classified into different types and vary between authors and countries. Niering (1985) identified nine types of wetlands for the United States based on hydrological conditions and vegetation, which include bogs, marshes, the Everglades, northern swamps, southern bottomland hardwood swamps, lakes and ponds, and rivers and streams. A wetland classification system for South Africa has been proposed by Dini et al. (1998) and is based on the system compiled by Cowardin (1979), which includes marine, estuarine, riverine, lacustrine and palustrine systems. Subsystems of the palustrine system include: flat, slope, valley bottom, and floodplain whilst riverine wetland systems include all wetlands contained within a channel (Dini et al., 1998). Channels have been defined as an open conduit, either natural or artificial, which periodically or continuously contains flowing water. Palustrine systems are vegetated wetlands traditionally called marshes, swamps, bogs, fens, and vleis. For the purposes of this study, riverine wetlands are of particular importance, due to their ability to act as natural filters of pollution conveyed in river channels. Wetlands occur primarily in landscape sinks, and as a consequence pollution flows into and collects in wetlands (Zedler and Kercher, 2005). Most of the wetlands in the Tshwane study area are riverine systems (Venter et al., 2005).

Wetlands provide a variety of ecosystem services such as ground water recharge, flood attenuation, pollution abatement, habitat for biodiversity, recreation opportunities, and provide for the livelihood of some communities through the harvesting of resources. The concept of valuing these ecosystem services has become widely accepted and specifically in urban areas people may care more about societal and economic benefits, rather than ecological integrity (Tong et al., 2006). The value of wetland ecosystem services within this context has been emphasised by Becker (1998), who states that wetlands are considered to be a good last line of defence for water quality problems. Wetlands can retain up to 96% of nitrogen and 97% of phosphorus and thus have the potential to significantly lower the cost of sewage treatment (Bolund and Hunhammar, 1999).

The loss and degradation of wetlands are a world-wide phenomenon with global wetland loss estimated at 26% (Moser et al., 1996). Within areas of heavy human development, the natural functions of wetlands can easily be disrupted (Mitsch and Gosselink, 2000). Tshwane is a large metropolitan area and wetland degradation has affected almost 84% of wetlands in the northern part of the municipal area (Venter et al., 2005). Roads, for example, are responsible for environmental impacts on almost 70% of wetlands in the aforementioned area (Venter et al., 2005). Alterations to wetland systems that lead to wetland degradation have been categorised by Brinson and Malvarez (2002) as geomorphic and hydrological disturbances, nutrients and contaminants, harvesting, extinctions and invasions, and climate change. The presence of pathogenic micro-organisms in surface water poses a health risk for water users, whether for recreational or domestic use (Pegram and Gorgens, 2001). Contamination of surface waters in South Africa has increased over the last decade and is linked to inadequate sanitation facilities due to rapid urbanisation. Given the ability of wetlands to improve surface water quality, the loss and degradation of wetlands is of particular concern.

Various studies (e.g. Turpie, 1995; United States Environmental Protection Agency, 2001; Greiner et al., 2005; Mitchell, 2005; Newbold, 2005; Giupponi and Vladimirova, 2006; Tsuzuki, 2006) have been conducted in different parts of the world, offering a variety of techniques to assess surface water quality and to prioritise areas for conservation or rehabilitation. By utilising and adapting current assessment and prioritisation techniques to the local situation in Tshwane, this study has investigated the opportunities for the City of Tshwane to prioritise the rehabilitation of wetlands on the basis of predefined ecosystem service objectives. The outcome of the study will be to maximise the attenuation of coliform bacteria in surface waters through targeted wetland rehabilitation projects in communities at risk from water borne diseases.

1.2 THE STUDY AREA

The City of Tshwane is situated in the Gauteng Province of the Republic of South Africa as indicated by Figure 1.1, covers a total area of 2 198 km^2 and includes the area north of the Magaliesberg range (Figure 1.2).

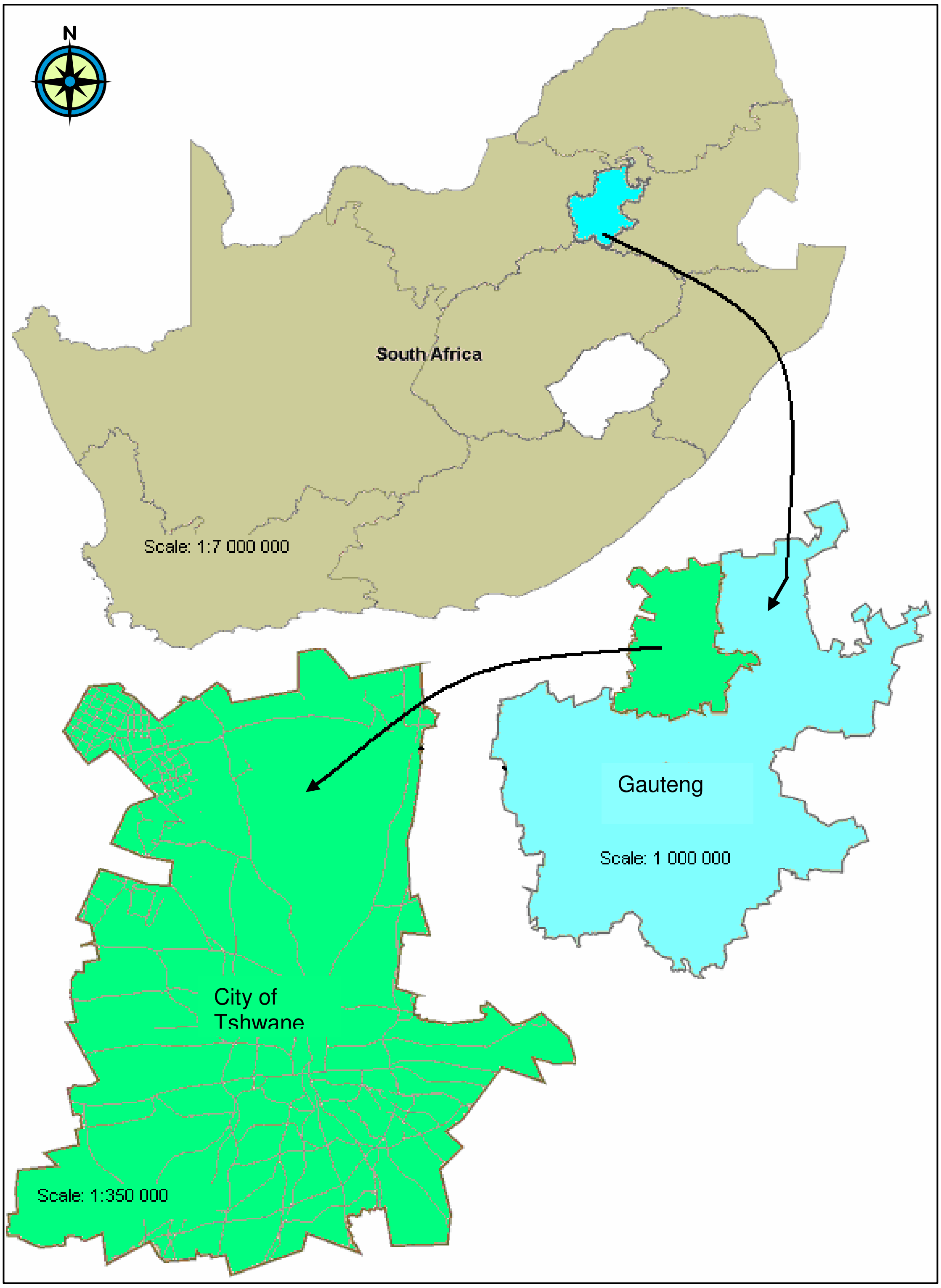


Figure 1.1: The location of the City of Tshwane in the Republic of South Africa and the Gauteng Province (City of Tshwane 2005a, 1-1).

The study area is situated in the Pienaars River catchment and four of its tributaries (Soutpan, Stinkwater, Apies and Sand Rivers) are included in the study. The climate can be regarded as warm to hot with moist to wet summers and dry winters. The long-term average rainfall of the total study area is 619mm per annum (Venter et al., 2005) but ranges between 600 and 800 mm per annum within the study area (Figure 1.3). The lower reaches of all the rivers in the study area receive rainfall ranging between 401 and 600mm on average per annum, while a part of the Apies River basin receives more rain – between 601 and 800mm per annum. According to Venter et al. (2005), 85% of annual rainfall occurs from October to March. Most of the study area is underlain by coarse-grained granite and granophyre of the Lebowa Granite Suite (Bushveld Igneous Complex), leading to shallow, coarse-grained, light textured soils and undulating topography (Venter et al., 2005). Two major soil classes are present in the study area – soils of restricted depth, seasonal wetness, and highly erodible and undifferentiated shallow soils, also highly erodible. The water- holding capacity of soils ranges from low to medium (Figure 1.4). According to the Department of Agriculture (2006) the classification of water-holding capacity as indicated in Figure 1.4, is a robust, explorative-level coverage that provides an overview for land assessment only. The map in Figure 1.4 is based on a distinction between stable and unstable soils. Apedal soils and soils with E horizons were grouped as stable, while strongly structured soils were classified as unstable (Department of Agriculture, 2006). The area is relatively flat, with gentle or low ridges (Figure 1.5) and consequently with a low potential for water induced soil erosion. The study area forms part of the savanna biome, as can be seen in Figure 1.6. Major veld types in accordance with Acocks Veld Types are bushveld and savanna, consisting of mainly Sourish Mixed Bushveld, Mixed Bushveld veld type, and Other Turf Thornveld (Venter et al., 2005). What is of importance for the study however is that the climate, geology and soil are fairly uniform across the study area. Palustrine wetlands constitute 7% of the study area, of which 50% are valley bottom wetlands, including three flood plain wetlands. The rest of the wetlands in the study area (47%) are seepage wetlands and the remaining 3% are pans (Venter et al., 2005). Only 16% of wetlands in the study area are near pristine, while 79% are degraded, and 5% are threatened (Venter et al., 2005).

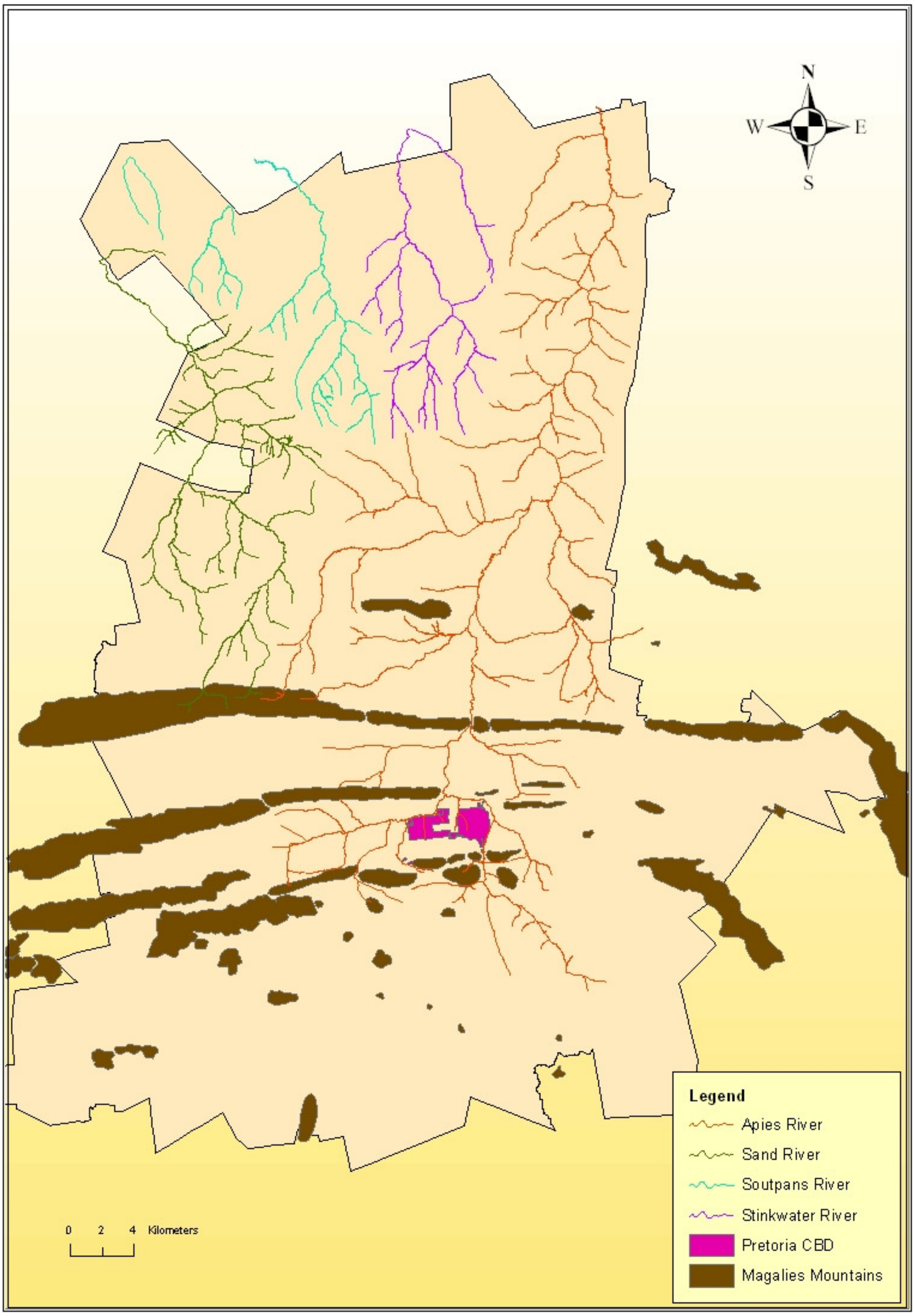


Figure 1.2: The study area.

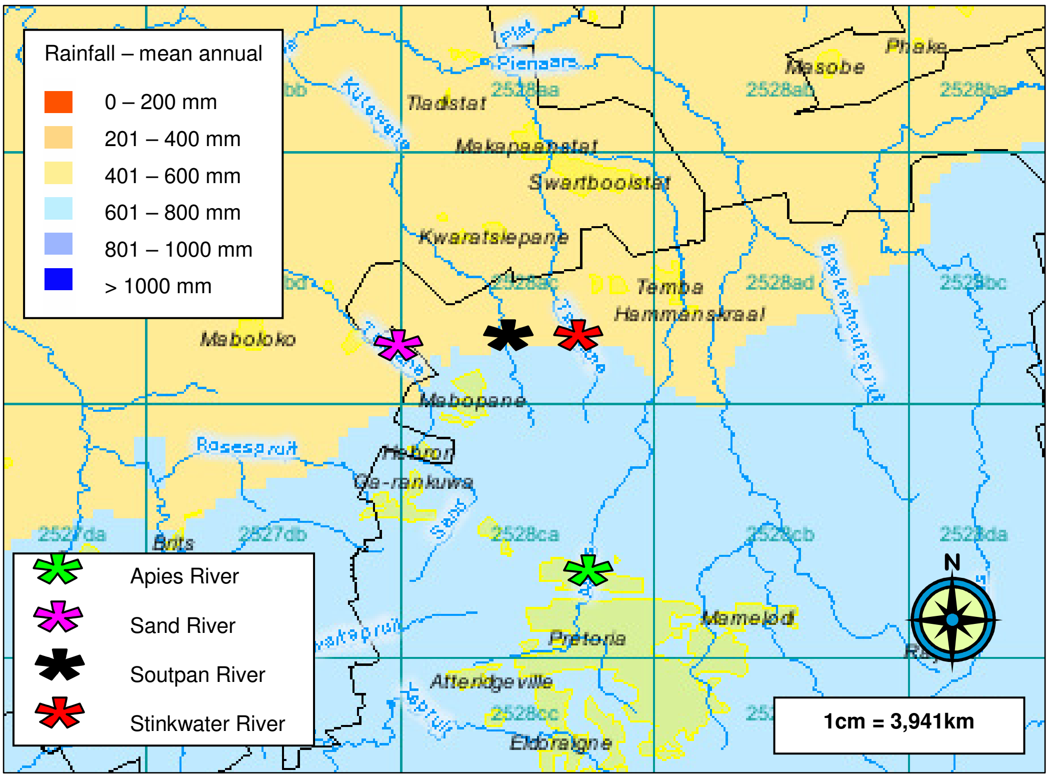
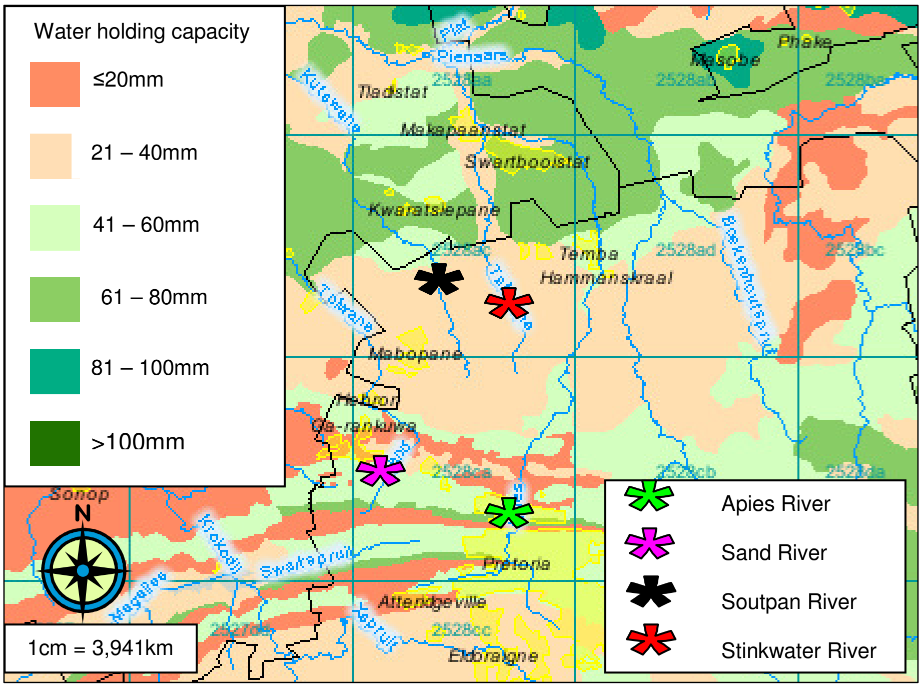


Figure 1.3: Mean annual rainfall in the study area (Department of Agriculture, 2006).



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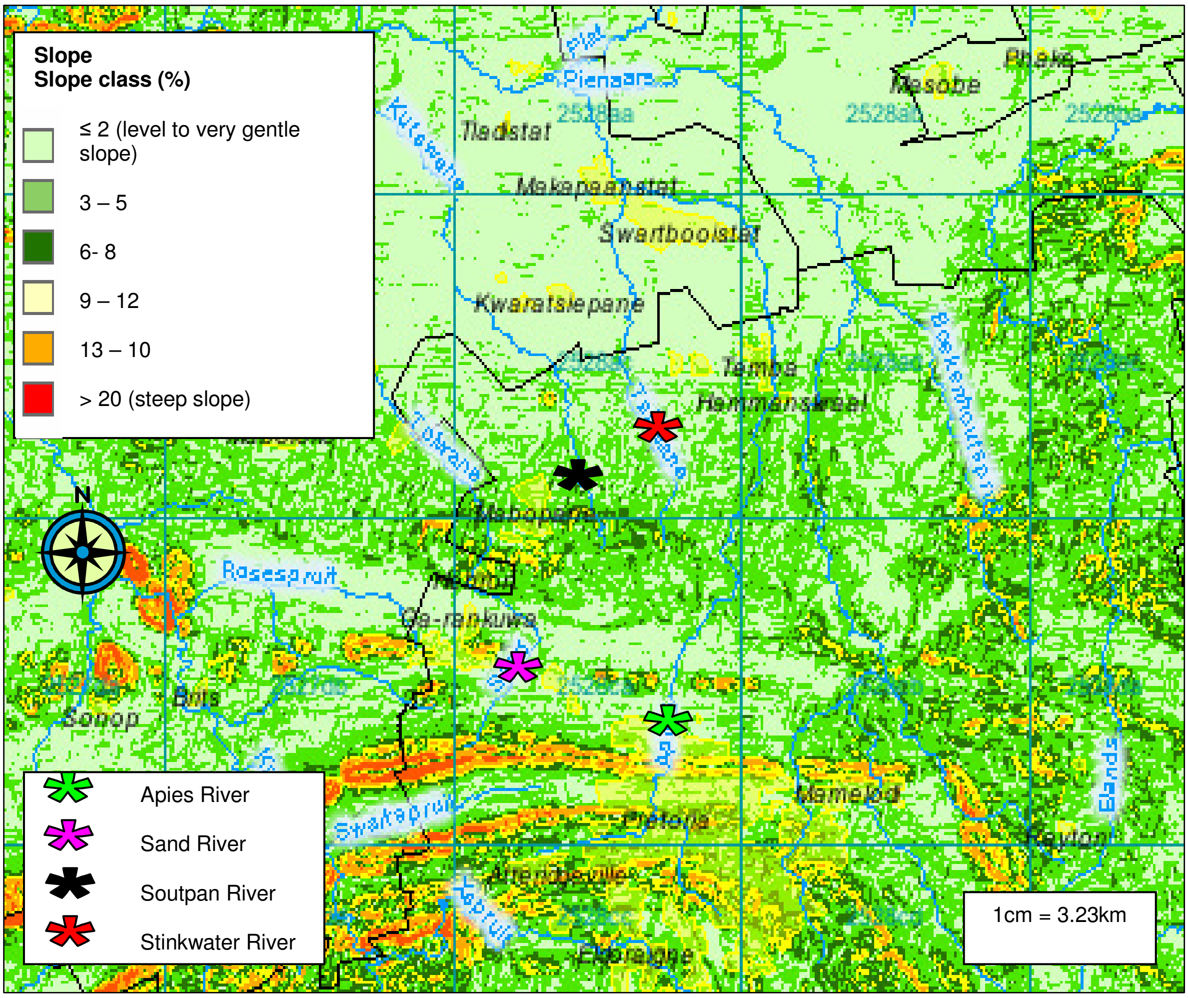

Figure 1.5: Slope classes of the study area (Department of Agriculture, 2006).

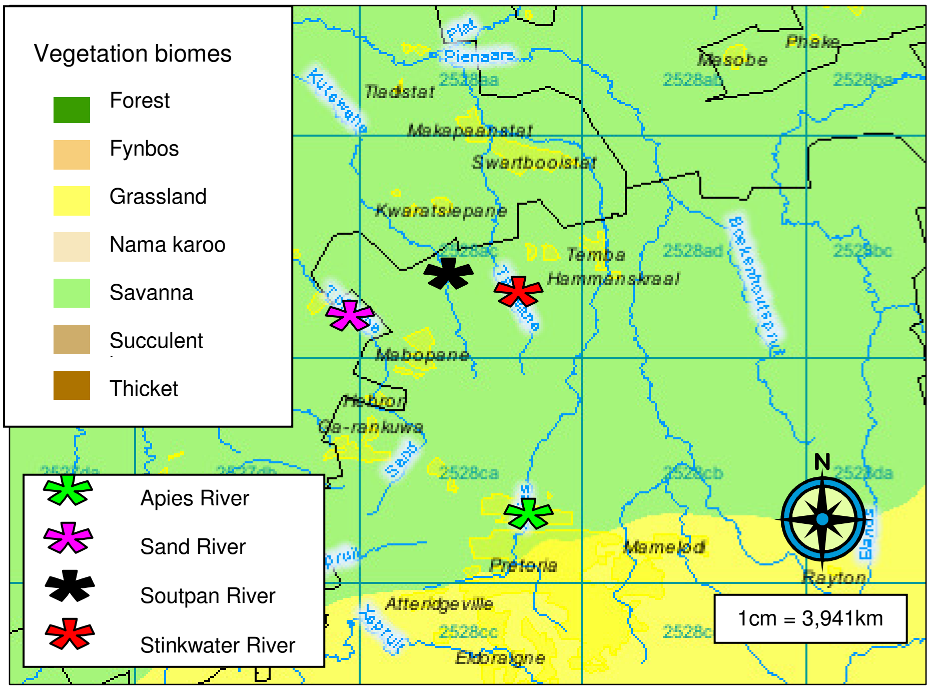

Figure 1.6: Representative vegetation biome of the study area (Department of Agriculture, 2006).

1.3 BACKGROUND

Due to a lack of information to evaluate the impact of planned development on wetlands, the City of Tshwane commissioned an inventory of wetlands during 2005 within the northern areas of its area of jurisdiction, which constitutes phase 1 of the complete project (Venter et al., 2005). A total of 192 wetlands were mapped and the conservation status of each was assessed. Figure 1.8 provides an overview of the main impacts on wetlands in the study area. Disturbance of the hydrological regime in the form of roads and dams are amongst the main threats to wetlands in the study area. The study was conducted by the Agricultural Research Council's Institute of Soil, Climate, and Water. Conservation status was classified as either pristine, degraded or threatened and the need for the rehabilitation of a wetland was indicated as either none, low, high, or urgent (see Table 1.1). According to Venter et al. (2005), 22% of wetlands have a high need for rehabilitation. Given limited funding for rehabilitation, it is now necessary to identify which of the 43 wetlands with a high need for rehabilitation, should receive priority attention.

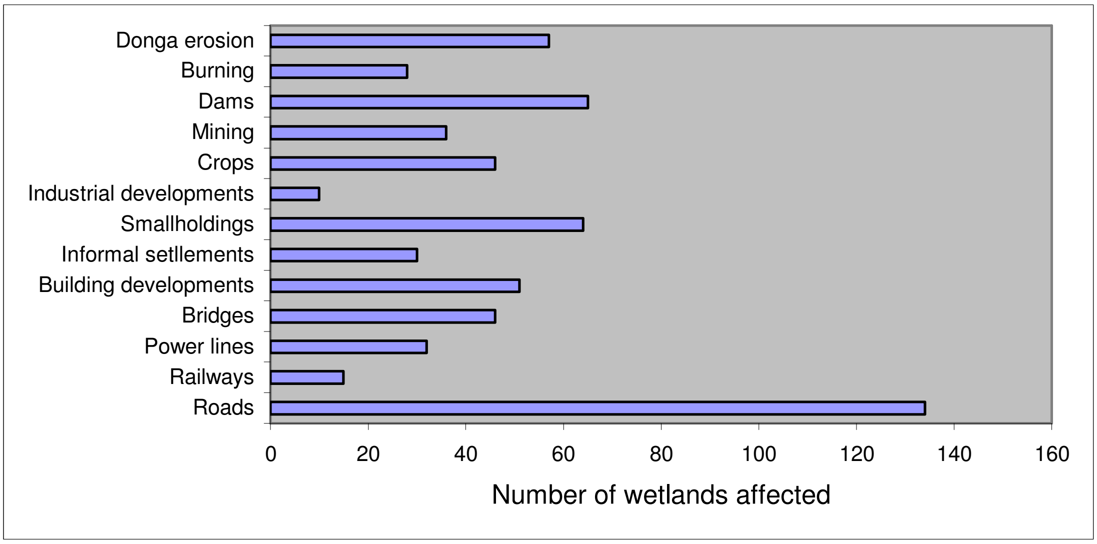

Figure 1.7:The threats to wetlands in the northern parts of Tshwane (after Venter et al., 2005).

High coliform counts in surface waters within the Tshwane area pose a risk for water-borne diseases, especially given the fact that only 46% of households in the Tshwane area of jurisdiction have access to piped water in their homes, and only 58% have access to sanitation (City of Tshwane, 2006). Levels of faecal coliforms in surface waters in the Tshwane area often exceed the standards for surface water quality by several magnitudes (City of Tshwane, 2006). Sample results from 2002 to 2005 show a year-on-year increase at some sampling points of the average faecal coliform counts (Table 1.2). The reasons for this are manifold and range from sewage pipe leakages, to the presence of settlements without basic sanitation.

Table 1.1: Conservation status and rehabilitation needs of wetlands in the Tshwane-north study area (after Venter, et al., 2005).

Conservation Status	Number of wetlands	Rehabilitation Need	Number of wetlands
Pristine	31	None	62
Degraded	151	Low	87
Threatened	10	High	43

Table 1.2: Periodic averages of faecal coliform counts per 100 mℓ at various sampling points in Tshwane's rivers (City of Tshwane State of the Environment Report, 2006).

Sampling point	May-Aug 2002	Sep-Dec 2002	Jan-Apr 2003	May-Aug 2003	Sep-Dec 2003	Jan-Apr 2004	May-Aug 2004	Sep-Dec 2004	Jan-Apr 2005
A02	350	7 000	86 500	3 075	77 500	46 750	33 333	44 000	44 000
A04	51 000	51 500	78 500	24 500	151 250	82 750	21 133	90 666	63 000
B12	4 250	5 806	20 250	1 450	8 000	49 333	1 066	17 000	28 000
B21	150	4 824	23 275	330	3 190	31 600	1 133	5 325	27 000
B19	550	10 500	23 800	510	2 475	72 000	3 766	3 800	18 600
B09	14 000	145 000	121 500	39 000	111 000	150 000	30 700	93 500	69 500
C24	No data	No data	No data	No data	No data	2 100	700	3 560	4 167
C33B	No data	No data	No data	No data	No data	69 000	10 233	1 450	11 533
E54	80	100	2 700	420	1 210	4 200	2 950	4 300	333
E62	335	27	3 775	120	465	3 300	60	700	600

Wetlands are known to positively influence water quality. Constructed wetlands can achieve a 90% removal efficiency of disease causing micro-organisms, as has been found by Shutes (2001). In addition, small constructed wetlands in the United States of America have been found to reduce faecal coliform and enterococci counts by more than two orders of magnitude (Hench et al., 2003).

This study has investigated the opportunities for the City of Tshwane to prioritise the rehabilitation of wetlands on the basis of predefined ecosystem service objectives. This long-term objective is to maximise the attenuation of coliform bacteria in surface waters through targeted wetland rehabilitation projects in communities at risk from water borne diseases. The study results will be used to direct wetland rehabilitation efforts of the City of Tshwane and will contribute to the development of a prioritisation framework for wetland rehabilitation. Much of the study, however, has focused on the prioritisation of the four river basins in the study area, based on a qualitative assessment of projected pathogen loading in the surface waters of each basin.

1.4 PROBLEM STATEMENT

Previously, policy makers frequently considered wetlands as wastelands, which have led to the degradation or destruction of many wetlands (Adger and Luttrell, 2000). This under-rating of wetlands for providing valuable ecosystem services is evident from the fact that nearly 84% of wetlands in the northern part of Tshwane are degraded or threatened (Venter et al., 2005). Of the 192 wetlands mapped in the inventory study, 130 are in need of some form of rehabilitation (Venter et al., 2005) (Table 1.1). Given the poor track record of wetland conservation in Tshwane, and the fact that wetland rehabilitation is generally costly, strong motivation will be required to obtain funding to invest in wetland conservation and rehabilitation. Even though the wetland inventory study conducted by Venter et al. in 2005, indicated some wetlands to be more important for rehabilitation purposes than others, the question still remains which of these should be rehabilitated first. The rehabilitation of the forty-three wetlands that have been identified as important for rehabilitation still remains beyond the financial means of the municipality. There are considerable pressure on local government finances for addressing service backlogs in communities. It is therefore evident that requests for funding wetland rehabilitation must be based on demonstrable benefits to communities. The excessive faecal coliform loading in surface waters in the Tshwane area, presents an opportunity to highlight the role of wetlands in surface water quality and consequently to motivate on this basis for funding wetland rehabilitation. The aim was to select suitable wetlands that can positively influence water quality and to prioritise these for rehabilitation. From the literature review (Chapter 2), it is evident that prioritisation should start at a landscape level where rehabilitation priorities are determined by water resource management goals (O'Hara et al., 2000; Kotze et al., 2001). A hierarchy of screening is proposed, starting most often with catchment or quarternary catchment levels. However, very little work has focussed on the modelling of pathogens on a landscape scale (Haydon, 2006; Elliot and Trowsdale, 2007). Pathogen loads in surface waters are the result of a large number of variables and consequently pathogen modelling is currently characterised by uncertainties (Dennis et al., 1997). However, most authors propose that the simplest methods should be used to assist with management decisions on a regional scale (Mitchell, 2005; Haydon, 2006).

If the aim of wetland rehabilitation in Tshwane is based partly on human health benefits, the following questions arise:

- How can different river systems in the Tshwane area be compared with each other to enable prioritisation of a specific river system without necessitating water sampling?
- How can current prioritisation and assessment methodologies as portrayed in the literature be adapted to the local situation in Tshwane?

- How can the various types of faecal water pollution sources be compared with each other, for example agricultural run-off and urban-run-off, without necessitating water sampling?
- Which river system contains the most faecal pollution sources and consequently poses the greatest risk to the various recipients or receptors of poor water quality?
- Which river basin contains the greatest number of households at risk from utilising contaminated surface water?
- Which river basin contains a combination of faecal pollution sources and users at risk of contaminated surface water that will warrant preferential rehabilitation of its wetlands?
- To which criteria must a wetland comply in order to be considered suitable for the improvement of surface water quality?

The study consequently aimed to adapt the available methodology to the local situation in Tshwane.

1.5 PURPOSE OF THE STUDY

The aim of this study was to compile a list of candidate wetlands based on a multi-criteria scoring technique on a landscape level for rehabilitation purposes in the northern areas of Tshwane. To accomplish this, a suitable methodology for the prioritisation of firstly, river basins and secondly, wetlands in the study area has been compiled. Candidate wetlands have been selected with the aim of optimisation through rehabilitation of ecosystem regulation services to attain a reduction in total coliform counts in down stream surface waters. Due to the fact that wetlands in the study area occur across the four different quarternary catchments, or river basins in the study area, a screening of the pollution sources within each catchment had to precede the actual wetland prioritisation process.

The study had the following specific objectives:

a. To compile a list of weighted criteria for each type of faecal pollution source based on existing datasets to enable a landscape level assessment and comparison of the river basins in the study area.
b. To determine the extent of faecal pollution sources in each of the river basins in the study area.
c. To determine the number of households at risk from utilising contaminated surface water in each river basin and account for the number of users in the prioritisation phase.
d. To score and prioritise the river basins in the study area, using a Simple Additive Weighting technique.
e. To identify a candidate list of wetlands for the purposes of further screening within the boundaries of the top priority river basin in the study area.
f. To recommend criteria for further investigations for a site level assessment of short-listed wetlands.

1.6 LIMITATIONS AND DELIMITATION

Potential weaknesses of the study lie in the unverifiable validity of the historic data that will be used. Data related to run-off and infiltration volumes of rainfall are not currently available for the study. The study is limited to the historical data that are available for the northern parts of the City of Tshwane. It was also difficult to predict faecal indicator loading rates from physical principles or values found in the literature.

CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

This chapter provides an overview of literature relevant for the study. An overview of important wetland concepts is provided, as well as information pertaining to microbial water quality, point and non-point source pollution, water quality modelling and the assignment of weights to variables used in water quality modelling parameters. Critique is expressed regarding certain viewpoints and contradictory findings by some authors are pointed out. Concluding statements are made regarding the need for a tailor made solution to wetland rehabilitation in the Tshwane area.

2.2 AN OVERVIEW OF WETLANDS

Wetlands are often described as the "kidneys of the landscape" because of their role in filtering the effects of run-off from surrounding land use, and have widely recognized functions that include storm water retention, shoreline protection, and wildlife habitat (Hunt et al., 1996). There is a worldwide recognition of the value of the ecological services provided by wetlands (Tong et al., 2006), and consequently the restoration of wetlands, which have in the past been considered as wastelands, have become a top priority. The alarming rate of global wetland loss has prompted amongst others, the development of the Ramsar Convention (Moser et al., 1996). Current global wetland loss is thought to be approximately 26% with different regions of the world displaying varying figures (for example 53% of wetlands were lost in the

United States between 1780 and 1980) (Moser et al., 1996). Zedler and Kercher (2005) however, estimate global wetland loss at 50%. Current estimations are that wetlands comprise only 9% of the global surface and that wetland loss and degradation leads to loss of a variety of ecosystem services, such as biodiversity support, water quality improvement, flood abatement and carbon management (Zedler and Kercher, 2005).

Wetlands and the classification of wetlands vary between nations, but in most countries vegetation type, hydrology, and topography are the major criteria for determining and classifying wetlands. The Ramsar Convention places emphasis on geographical location, hydrological conditions, and vegetation type and includes both natural and man-made wetlands (Moser et al., 1996). Criteria of the Cowardin-based classification include substrate materials, flooding regime or vegetation life form, while in Germany the chemical properties of water, the source of water, the water level of groundwater and surface water are important. In terms of the hydrogeomorphic (HGM) classification, vegetation is of lesser importance (Kim et al., 2006). Instead, the HGM classification is considered better from a functional point of view (Kim et al., 2006). In the United States of America, the wetland classification has mostly focused on the appearances of the wetland types rather than their functions. Recently, however, attention has shifted to emphasize the evaluation of the impact of proposed projects on wetland functions (Kim et al., 2006).

The importance of wetland functions is to some extent acknowledged in South African legislation. The National Water Act (South Africa, 1998) includes wetlands as a type of watercourse that in turn is an integral part of water resources. The law provides for the protection of water resources and recognizes the interconnectedness of the physical and biological aspects of water systems and acknowledges that aquatic ecosystems are the resource base from which water is derived, rather than a competing water user. Wetlands are usually intricately linked with a network of streams and rivers in a catchment, and have a large impact on quality, quantity, and timing of stream flows. Dickens et al. (2003:6) state that wetlands "are a vital part of the water resources and that the wise use of wetlands provides a vehicle for promoting the wise and integrated management of catchments", and consequently may not be irreversibly damaged. South Africa only recently completed the first phase of a national wetland inventory – a vital prerequisite to ensure the protection of wetlands (Figure 2.1). Whilst wetland locations and wetland sizes have been mapped, wetlands have not been classified. On completion of the national wetland inventory project, a clearer picture will emerge of the diversity of South Africa's wetlands (South African National Biodiversity Institute, 2006). Preliminary results indicate that wetlands cover 15.3% of the South African surface area, with an average number of 62.7 wetlands per quaternary catchment (South African National Biodiversity Institute, 2006). The scale of Figure 2.1 is approximately 1: 19 178 660, and the GIS generated map is therefore not a visual representation of actual wetland coverage percentage mentioned above. National wetland rehabilitation initiatives are undertaken by Working for Wetlands – a programme sponsored by the National Department of Environmental Affairs and Tourism (South Africa, 2002), in an effort to provide employment through the rehabilitation of wetlands of national priority.

The recognition of wetland functionalities led to the economic valuation of wetlands and has become common practice as well as the focus of many studies. For example, a study in Sweden found that wetlands might significantly lower total abatement costs of sewage treatment plants (Byström, 2000). Because of specific hydrological, hydrochemical and ecological conditions, wetlands have a high potential for nutrient retention and nutrient transformation (Kieckbusch et al., 2006), and consequently play an important role in the development of sustainable water management strategies.

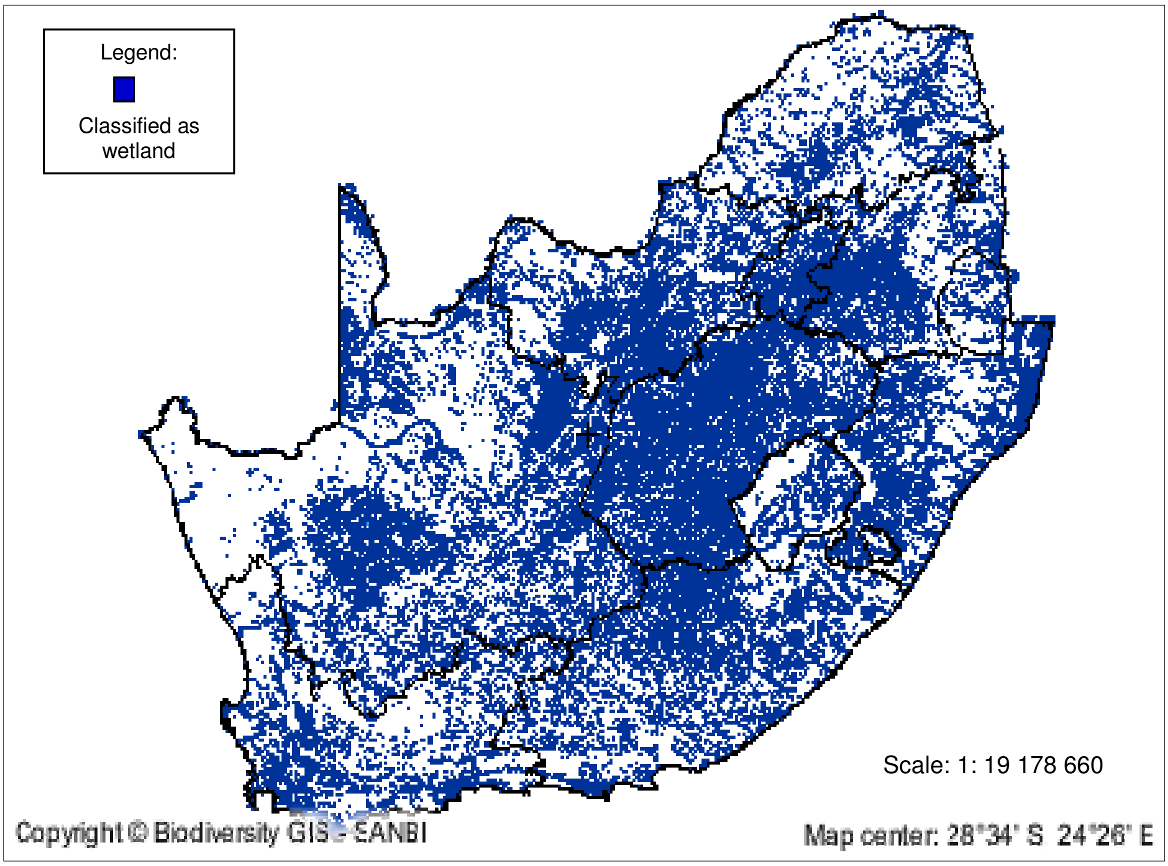

Figure 2.1: National wetland inventory of South Africa (South African National Biodiversity Institute, 2006).

Wetlands are effective for the removal of pollution from water to such an extent that the creation of artificial wetlands for pollution abatement is now accepted practice (Decamp and Warren, 2000; Shutes, 2001; Barnes et al., 2002; Merlin et al., 2002). For example, research on an artificially created coastal wetland ecosystem in the United Kingdom have found a 97% reduction of faecal indicator organisms in tidal cycles, wich has resulted in a significant improvement in bathing water quality (Kay et al., 2005). As such, it makes economic, social, and environmental sense to ensure the protection of naturally occurring wetlands.

Artificial wetlands are constructed to recreate, as far as possible, the structure, and function of natural wetlands (Shutes, 2001; Interstate

Technology and Regulatory Council, 2003; United States Environmental Protection Agency, 2005). This includes amongst others, the naturally occurring microbial community that affects the biochemical transformation of pollution, the simultaneous sedimentation, and filter effect created by plant communities and soil structure, as well as the high biotic productivity of wetlands. The nutrient removal functioning of wetlands is however, still not completely understood (Zedler and Kercher, 2005). It is also not clear where wetlands should be positioned in the landscape to improve water quality. Zedler and Kercher (2005) propose that preserving and restoring wetlands to improve water quality requires a landscape approach, in order to find sites that can intercept a significant fraction of the water shed run-off. Riverine wetlands, for example, due to their extensive interface with uplands, have the capacity and opportunity for improving water quality (Brinson, 1988). The importance of a landscape approach as preceding site-specific decisions on wetland management is also emphasised by Ethridge and Olson (1992). According to Nakamura et al. (2005), ecosystem-based management in the early 1990s was proposed to protect biodiversity, emphasizing the maintenance of ecosystem processes at larger scales. Modern restoration therefore acknowledges the importance of ecosystem patterns and processes occurring at landscape scales. Consequently, a hierarchical approach to rehabilitation and restoration is proposed by Nakamura et al. (2005). A mass-screening process should form part of such a hierarchical approach where spatial analyses at regional, catchment, and local scales are used to assess ecosystem patterns and processes in the context of restoration planning. This provides a clearly defined process of identifying rationales for, and measures of, restoration. The regional context is used to identify the ecosystem elements or functions requiring preservation or restoration. The Tshwane wetland prioritisation study will also follow a hierarchical approach, by firstly investigating which river basin in the study area is most heavily polluted, in order to optimise on a local scale, the relevant ecological services rendered by wetlands.

2.3 MICROBIAL WATER QUALITY

Urbanization is one of the most significant demographic trends of our recent times, and growth is particularly accelerating in lower-income countries. The majority of urban growth is associated with the rapid expansion of smaller urban centers and peri-urban developments. Unfortunately, much of this growth is unplanned and informal (Parkinson and Tayler, 2003), leading to poor sanitation conditions and a lack of safe drinking water. Poor sanitation often leads to the presence of pathogenic microorganisms in surface water and poses a health risk for water users, whether for recreational or domestic use (Pegram and Gorgens, 2001). Using contaminated water for irrigation, recreational or domestic use poses certain health risks associated with water borne diseases (Murray et al., 2004). A cholera epidemic in South Africa in 2000/2001, caused 265 mortalities and morbidity levels totalled 117 147 in five provinces (Van Vuuren, 2006). Research into the causes and outcomes of the epidemic concluded that: “the water-related disease threat for vulnerable communities is far from over” (Van Vuuren, 2006). Microbial pollution of water is measured by means of indicator organisms, rather than measuring the pathogens themselves, due to the large number of pathogenic organisms that can be present in surface water. Faecal coliforms is a widely used indicator organism for faecal pollution of water and has been defined as thermotolerant (maximum 44.5^0C) bacteria derived from the intestines of warm-blooded animals, including humans (Murray et al., 2004). For water to be considered potable, faecal coliforms must not be present (Murray et al., 2004). Microbial water quality guidelines for raw drinking water supply in South Africa have been set at 20 000/100ml (Venter et al., 1996), yet in Tshwane the average microbial concentrations in surface waters often exceed the guidelines (City of Tshwane, 2006).

South Africa (1999) has determined categories of non-point source water pollution, and pathogen contamination of streams has been indicated as a top priority on the basis of known extent and severity of impact. Contamination of surface waters in South Africa has increased over the last decade and is amongst other reasons linked to inadequate sanitation facilities due to rapid urbanisation. A study conducted by Venter et al. (1997) in a peri-urban catchment in South Africa found that discharges from sewage plants and run-off from informal settlements were the main factors affecting microbial water quality. The significant role of effluent from wastewater treatment plants on surface water quality was also recorded by Momba et al. (2006), who concluded that reliable water treatment processes in developing countries are the exception rather than the rule. Land-use, which has been highlighted by Conway and Lathrop (2005), is another important source of non-point source pollution - a figure as low as only 10% of impervious surface cover would impact negatively on surface water quality. Mallin et al. (2000) also indicated that the most important anthropogenic factor associated with faecal coliform abundance in watersheds, is percentage watershed impervious surface area. These surfaces, consisting of roofs, roads, driveways, parking lots etc., serve to concentrate and convey storm water borne pollution to surface waters. Linear regression analysis indicated that percentage watershed-imperviousness alone could explain 95% of the variability in average faecal coliform abundance (Mallin et al., 2000). Animal feeding operations (United States Environmental Protection Agency, 1998; Boyacioglu, 2006; Giupponi and Vladimirova, 2006) were found to be significant sources of non-point pathogen sources to surface waters. A study by the United States Environmental Protection Agency (1998) found that agriculture was responsible for 70% of water quality impacts on rivers. The land-application of manure and sewage sludge were also found to be significant sources of pathogens - even after typical wastewater treatment, significant numbers of pathogens remain in sewage sludge that could contaminate surface water bodies (Gerba and Smith, 2005). The relative contribution of these typical sources differs during low and high-flow conditions, according to Boyacioglu (2006). Agriculture was found to be the dominant contributing factor during low flow conditions, whilst urban land-uses (or non-point sources), were the main contributor of pollution during high-flow periods. For the purposes of the study, no distinction will be made between flow conditions, as the study is concerned with the cumulative effect of various pollution sources on surface waters.

Pathogens can be transported for significant distances in ground and surface waters whilst overland transport of pathogens can be, according to some authors, significantly attenuated by vegetated surfaces (Gerba and Smith, 2005). Depending on rainfall intensity and vegetation strip length, recovery percentages of *Cryptospordium* range from 0.8 to 27.2% (Trask et al., 2004). Similar findings were recorded by Tian et al. (2002), who found an inverted association between faecal pollution delivery ratio and distance between source and stream. A range of between 0.06 and 0.182 die-off coefficients for the attenuation rates for *E. coli* was used, which is required by the model for spatial and temporal variation rates (Tian et al., 2002). Different die-off coefficients however, are utilised by the United States Environmental Protection Agency (Table 2.1). It is also mentioned by Tian et al. (2002) that microbes within bovine faecal matter may remain on soil and plant surfaces for up to 1,5 years. Bacterial retention potential of soils is influenced by a large number of variables such as physical, chemical, and structural properties of the soil layers. Nola et al. (2006) found ranges of total coliform detention in soil from 69.22% to 99.95% during a study to determine the groundwater retention potential of soils above the ground water table in the Central African region. The reason for the variation in retention potential was concluded to be due to numerous and variable physical, chemical, and structural properties of soil layers, as well as the interactions between soil layers and bacteria (Nola et al.,2006). It can therefore be expected that on a landscape scale, soil retention of bacteria will vary considerably, yet it is evident that large numbers of pathogens will eventually find their way to surface waters during storm events. The main pathway for faecal matter of agricultural origin to streams and rivers is through surface water run-off (Tian et al., 2002). Various other studies also found a correlation between pathogen load and run-off (Haydon, 2006). Pathogens deposited on the catchment area, acts as a pathogen store. Pathogens are mobilised during rainfall events and transported in run-off (Haydon, 2006). This is further supported by Venter et al. (1997), who in a series of modelling runs predicting faecal coliform levels, demonstrated that bacterial die-off did not result in significant improvement of the microbial water quality of the surface water in the catchment investigated. Die-off rates for faecal bacteria in surface waters have been established by the United States Environmental Protection Agency (2001) as 0.15 $K_B(day^{-1})^a$, (where ${}^aK_B$ is the overall first-order decay rate) but other studies (Venter et al., 1997; Tian et al., 2002; Haydon, 2006) found viable bacteria in river water after extended residence periods. Given the risk-based nature of the study, it is expected that households at risk of contaminated surface water, still receive very large numbers of viable faecal bacteria due to extended residence periods of some faecal bacteria.

Various studies found a significant correlation between turbidity and enteric bacterial abundance as well as increased survival rates of faecal bacteria when associated with sediments (Mallin et al., 2000; Pegram and Gorgens, 2001). Re-suspension of sediment can cause re-release of bacteria into water. Mallin et al. (2000) found a highly significant proportion of faecal bacteria associated with particulate matter in water columns. However, faecal bacteria display great variances in survival rates. For instance, *E. coli* 0157:H7 was found in water after 91 days, *Salmonella species* remained active in surface water after 152 days and viable *Yersinia enterocolitica* was found after 64 weeks (Guan and Holley, 2003). In comparison, survival rates of bacteria in soil were recorded to be 8 weeks for *E. coli*, 63 days for *Salmonella* and 8 weeks for *Cryprospordium* (Guan and Holley, 2003). Survival rates of bacteria are influenced by temperature, soil moisture, ultra-violet radiation, and available nutrients (Venter et al., 1996).

Data for the purpose of this study on source distance from surface waters are not known, with the exception of waste water treatment works, which release effluent directly into surface waters. Given the possible variances of soil types within a given surface area, inadequate data are available for the purposes of detailed modelling of soil retention and die-off rates in this study. As such, allocating relative weights to the different sources will be based on assumptions related to die-off rates and soil retention. The same set of assumptions will be used for the entire study area.

Table 2.1: Faecal indicator die-off rates (after United States Environmental Protection Agency, 2001).

Environment	Die-off coefficient of faecal coliforms
Sand (at 25 ◦C)	0.124
Loam (at 25 ◦C)	0.108
Clay (at 25 ◦C)	0.022
Non-sterile river water (12 days)	0.15
Sediment (at 8 ◦C)	0.010 – 0.023

2.4 POINT AND NON-POINT SOURCE POLLUTION

The presence of sediment and pathogenic bacteria in surface waters are the result of point and non-point source pollution. Non-point source pollution represents land-use areas and activities that result in the mobilisation and discharge of pollution in any matter other than through a discrete or discernable conveyance (Pegram and Gorgens, 2001). Point source pollution is generally referred to as: ”discernable and confined sources of pollution that discharges from a single conveyance, such as a pipe, ditch, conduit or channel” (Pegram and Gorgens, 2001, 7). Land-use is generally assumed to be the overriding determinant of water quality impacts (Pegram and Gorgens, 2001). Informal settlements, for example, are a land-use type contributing significant amounts of pathogens to surface waters (Jagals et al., 1995).

A conceptual framework for water quality aspects arising from point and non-point sources has been compiled by Pegram and Gorgens (2001), who postulated a continuum of production at source, delivery from the source to surface waters and transport of pollution through the water medium to water users. Production includes the mobilisation and attenuation of pollution. All of these aspects, together with many variables involved in each mechanism, result in the specific quality of a surface water body, and the requirements set for water quality. Non-point source pollution assessment is guided by management goals, water quality concerns, and source area character. Principles for the selection of an appropriate assessment technique should, according to Pegram and Gorgens (2001), be guided by amongst others a clear statement of management goals, the use of the simplest technique that will provide the required information using a technique that matches time, budget and data availability.

It has also been proposed by Pegram and Gorgens (2001) that the assessment process of non-point source pollution combine qualitative assessment, data analysis, and modelling where possible, as no single assessment will provide the whole picture. This will enable a process that is iterative and becomes progressively more focussed over time. Furthermore, the attainment of adequate results that determine the closure of the analysis should be based on the law of diminishing returns. This implies that for pollution such as pathogens, qualitative assessment and identification of sources with high production rates are the most appropriate high-level techniques. This is in contrast with most studies undertaken on surface water quality, which tend to divide catchments or areas under assessment into grids, in an effort to include the complete range of aspects involved in non-point source pollution. A large number of variables for that particular grid are added to the results from other grids by means of considerable computational power (Foster and McDonald, 2000; Hassen and Prou, 2001; Newbold, 2005; Mitchell, 2005; Giupponi and Vladimirova, 2006; Liu et al., 2006; and Tsuzuki, 2006). However, management decisions for surface water are still made under considerable uncertainty. Few management tools are available that calculate uncertainty interactively and therefore deterministic non-point source pollution models are commonly used as management tools for evaluating best management practices for watersheds containing significant non-point sources. The simulation of non-point source pollution models requires large amounts of data and is a limiting factor. Although methods have been developed for assessing uncertainty in non-point source pollution models, decision risk is commonly handled by performing 'off-line' simulations – using GIS.

Shortcomings of this approach include the possibility that simulated loads might not correspond with actual loads, due to a number of factors, such as surface attenuation and bacterial die-off. Further critique against most non-point source models is that the water shed or river basin under investigation is typically divided into hundreds of fields or grids, depending on its size, and since the number of units in the network can be very large, the method can become quite cumbersome, costly and time consuming. Using discrete units for a process that presents in essence continuous values, can also be problematic, unless an adequate number of units are introduced. Studies based on this methodology include those by Greiner (2005), Newbold (2005), Giupponi and Vladimirova (2006) and Park et al. (2006). The inherent uncertainty of these models is further highlighted by Dennis et al. (1997), according to whom the single greatest challenge to the modelling of non-point source pollution is to obtain sufficient data to characterize the temporal and spatial distribution of this type of pollution with knowledge of its uncertainty. They mention that soil, for example, is an extremely heterogeneous medium with considerable spatial variability regarding many of its properties. The physical, chemical, and biological properties influencing the fate and movement of non-point source pollution can consequently vary greatly over short distances (both laterally and vertically) and often vary independently to one another. This has a significant influence on the transport processes of non-point source pollution. Consequently, input data and values of parameters used in the modelling of non-point source pollution is its greatest limitation.

Potential maps are another technique for the assessment of non-point sources and provide a spatial indication of the relative availability of a contaminant (Pegram and Gorgens, 2001). This is based on land-use activity-related application and removal rates. Sediment and microbiological availability are estimated, based on simple representation of the production mechanism associated with land-use in the area. Potential maps are usually compiled by overlaying or combining a series of spatial data by means of a GIS. Key aspects that affect contaminant generation, application, removal, and/or assimilation are incorporated. Potential maps consequently only provide an indication of the potential impact and not the actual yield. Compilation of potential maps is amongst other aspects, based on spatial GIS coverages of land-uses. For example, potential maps were used in the Mgeni catchment to estimate soil loss potential and identify land-uses and areas for further management attention (Kienzle et al., 1997). GIS coverage of twenty one land cover classes were used to provide a spatial indication of areas with high and low soil loss potential. Hazard maps on the other hand, indicate impacts associated with non-point source pollution and include the delivery aspect of the conceptual model. Assessments incorporating the delivery processes of contaminants to surface water, such as surface wash-off, interflow and groundwater flow are less reliable due to the large number of variables involved, and as such potential maps are more often used. Furthermore, according to Pegram and Gorgens, (2001) pathogens die off in a matter of days, and thus daily production rates are used for the generation of potential maps.

Potential maps are a more cost-effective and a less data intensive method. Since the method is based on the fact that various land-uses are associated with different levels of contaminants, which are transported to surface water during rain events, unit area loads can provide estimates of average pollution loads for catchments. This is done by utilising empirical estimates of the mass of pollution exported per unit area per unit time for a particular land–use (Pegram and Gorgens, 2001). Typical areal production rates of *E.coli* (x 10^6/ha/day) for general land uses have been given by Pegram and Gorgens (2001) in Table 2.2.

The two most highly urbanised of the six sub basins assessed by Kappel et al. (undated) in New York State, had the highest loads of all constituents measured. For example, the Total Suspended Solid (TSS) yield for the high-density residential area was found to be 512 kg/km^2/day, almost 6.5 times more than the rural areas, which yielded 79,7kg/km^2/day of TSS. The findings by Kappel et al. (undated) thus support the findings by Pegram and Gorgens (2001) as the range of probable contaminant yield increase from rural to urban areas as indicated in Table 2.2. Sediment and faecal coliform concentrations generally have a linear relationship (Pegram and Gorgens, 2001 and Mallin et al, 2000).

Potential sources of microbial contamination have been indicated as dense settlements, areas lacking in sanitation, intensive livestock farming as well as contaminated discharge from waste water treatment works (Murray et al., 2004). Despite the fact that many of these sources are not directly linked to surface waters, Guan and Holley (2003) state that intensive livestock farming and leaking sewer pipes can be a significant source of faecal bacteria, as significant numbers of faecal bacteria could reach surface waters by infiltrating and travelling through soil and subsurface tile drains.

Park et al. (2006) describe a process in which a stream network is divided into sub domains, with a similar number of tributaries for the purposes of designing a water quality network to detect pollution sources implicated in water quality changes. In this method, each tributary is treated as a pollution source. A river reach is formed by the intersection of two upstream tributaries and the fluvial magnitude increases with flow distance, reaching a maximum value at the river mouth, which represents the total number of tributaries. Samples taken at this point will represent the total pollution load of the river system.

For the purposes of ranking river basins for prioritising conservation efforts, pollution load is expected to reach its maximum in the lowest reaches of a basin, where, in the case of the study area, most informal settlements are located. Thus when ranking river basins, only the total projected pollution load will be considered.

Table 2.2: Typical areal production rates of *E.coli* (x 10^6/ha/day) on general land uses (after Pegram and Gorgens, 2001).

Rural	Informal	Urban
10 – 50	50 – 500	1 - 100

Since poor water quality is a concern to its users, the ranking of river basins must include an investigation into the users in the catchment. Sensitive water users have been indicated as those drinking untreated or partially treated water, having partial or full contact with surface water or irrigation with surface water and eating raw crops (Murray et al., 2004). Although various other authors recorded similar findings (Coit, 2002; Keraita et al., 2003; Parkinson and Tayler, 2003), this study will include only those users drinking untreated or partially treated water.

2.5 WETLAND CLASSIFICATION AND WETLAND PRIORITISATION RELATED TO FUNCTION

A classification of the hydrogeomorphic settings supporting inland wetlands has been used to indicate the water resource function of different types of wetlands (Dickens et al., 2003). This included hillslope seepage wetlands, important for the regulation of the flow of shallow groundwater which is important in binding soil and preventing soil erosion. Both valley bottom wetlands with and without a channel, slow down floodwaters, trap sediments, remove nutrients and pollution from upstream sources, as well as prevent soil erosion. Depression wetlands retain nutrients and re-charge groundwater.

Valley bottom wetlands with or without a channel are of particular importance for the purposes of this study, due to their ability to remove pollution from upstream sources. This is due to their extensive interface with the larger catchment and the direct contact with large quantities of surface water runoff, either in the flood plain or in the channel (Brinson, 1988). Wetland prioritisation implies the ranking of wetlands on the basis of the functions that wetlands are expected to perform in a landscape (Kotze et al., 2001). These functions have been described by various authors and have been summarised by Dickens et al. (2003) as water supply, flow regulation, erosion control, sediment removal or detention, nutrient removal or detention, the removal of toxicants, providing conservation, tourism and recreation opportunities. It is thus possible to prioritise wetlands on the basis of their functionalities and certain pre-defined objectives related to the management of surface waters or catchments.

It is important to note that the value of wetlands depends on their hydrogeomorphic position in the landscape and the position of human settlements who find value in these ecosystems (Mitsch and Gosselink, 2000). However, wetland restoration efforts are seldom as successful as envisaged, which is often due to a lack of consideration of landscape factors affecting wetlands and their catchments (O'Hara et al. 2002). In most cases the prioritisation of surface water problems such as non-point source pollution or river/wetland conservation and rehabilitation starts with prioritisation and planning on a catchment or a quaternary catchment scale (Kotze et al., 2001; Quin, 2003; King et al., 2003; Murray et al., 2004), based on the realisation that catchment-level processes affect local form and function. Prioritisation on this scale is also more cost effective according to Murray et al. (2004). Short listed catchments for prioritisation have been indicated as catchments with microbial water quality problems, high incidences of water-borne diseases, untreated or partially treated water used by local households, and those with settlements that do not have adequate sanitation infrastructure (Murray et al., 2004). The prioritisation criteria suggested by Murray et al. (2004), are clearly based on optimising wetland functionalities, an approach which originates from Dickens et al. (2003). As a first step, it is proposed that areas of the catchment where more detail is required should be prioritised for an inventory and consequent action (Dickens et al., 2003). Criteria for such a prioritisation exercise are thus similar to that proposed by Murray et al. (2004), but include representative habitats for unique biodiversity preservation and sustaining of livelihoods. A process for wetland rehabilitation in South Africa for the purposes of rehabilitation has been described by Kotze et al. (2001). The process is based on strategic objectives for wetland rehabilitation. This is followed by the identification of priority catchments, based on critical water quality. This is followed by the identification of priority quaternary sub-catchments by utilising criteria such as land-uses and the opportunity for the creation of employment. The proposed prioritisation process is also based on optimising wetland functionalities and the role such wetlands would play in a catchment (Kotze et al., 2001). Prioritization of individual wetlands for rehabilitation is based on the opportunity and potential to perform certain hydrological functions, as well as the costs associated with rehabilitation.

The prioritisation of wetlands based on their functionalities, forms the basis of many prioritisation criteria, as is evident from the discussion thus far. Whether rehabilitated for water quality improvement or biodiversity conservation, wetland functionalities can be overwhelmed by the sheer extent of disturbances in their catchments; wetland rehabilitation therefore has to take place within a broader framework of catchment management. The ability of a wetland to provide an ecosystem service such as water quality improvement, is reflected by landscape variables related to upstream and downstream land uses and land configurations (King et al., 2003), and therefore wetlands should not be viewed as isolated systems in the landscape, but rather as a response to broader environmental patterns. Wetlands are too often perceived as isolated from the larger ecosystems from which they form a part, whether it be aquatic or terrestrial. It is also proposed that total wetland surface area required for optimising wetland functionalities, should be determined. Riverine wetlands are furthermore both sources (if degraded) and sinks of pollution, and should therefore be considered as a network of wetlands, rather than isolated patches in the landscape.

In order to optimise functionalities related to water quality improvement, individual wetlands within a priority catchment area should ideally also be screened for specific characteristics. The characteristics of a wetland that are most effective in decreasing suspended solids have been described by Brown (1985). This includes an impoundment or detention of runoff to increase sedimentation, an undefined inflow channel into the wetland, resulting in better dispersion of incoming sediments, and a dense vegetative growth throughout the wetland to reduce flow velocity and wave action. The prolonged interaction of water with wetlands facilitates the biochemical transformation of some water-quality constituents (Suurballe, 1992). The importance of direct contact between surface waters and wetlands has been recognised by The United States Environmental Protection Agency (2005). Wetlands bordering first-order streams were found to be effective for the removal of pollution, including sediment from surface waters due to a smaller percentage of total stream water coming into contact with the wetlands (United States Environmental Protection Agency, 2005). Close proximity of wetlands to the source of non-point pollution, optimises wetland functionalities (United States Environmental Protection Agency, 2005). A report on strategies for assessing the cumulative effects of wetland alteration on water quality concluded that riverine wetlands are more important than fringe wetlands, which are small in comparison with the volume of water that flushes through them (Brinson, 1988). Wetland systems that have long hydraulic residence times allow for improved sedimentation of solids and it is estimated that 80% of total suspended solids are removed in the first 2 days of the theoretical hydraulic retention time (United States Environmental Protection Agency, 1993). Literature reviewed is mostly consistent on the importance of direct contact and residence time. With reference to vegetation, wetlands with dense stands of vegetation enhance sedimentation through decreasing flow rates (United States Environmental Protection Agency, 1993).

A study by Kay et al. (2005) recorded a reduction of microbial flux to a coastal bathing area by a wetland consisting of an area covering 146.4 ha or 1.56% of the total area draining through it. The wetland reduced microbial content of the bathing area by up to 97%. It was estimated that 1% to 5 % of the total watershed area of the Des Plaines River in Illinois needs to consist of wetlands to improve water quality and up to 15% of the surface area for the Great Lakes Basin in Michigan, United States of America (Zedler and Kercher, 2005).

Areal removal rates for pollution have been used as a “rule of thumb” in the sizing of constructed wetlands for the treatment of municipal wastewater; where removal is expressed as mass removed per unit area of wetland using the following formula (Interstate Technology and Regulatory Council, 2003): Areal removal Rate = gm/m^2. For constructed wetlands, the pollution load in the incoming waste stream and the desired end quality determines wetland size. Depending on the concentration of a specific pollution in wastewater, a specific wetland size is consequently recommended – greater pollution concentrations will require a wetland larger in size. Large wetlands are expected to be more efficient for the improvement of water quality than small wetlands due to their relative increases in retention time. Table 2.3 presents the typical size distribution of constructed wetlands in the United States where the majority of constructed wetlands are less than 10ha. Minimum width has been indicated as 61m and minimum length as 15m (United States Environmental Protection Agency, 2000). For the purposes of this study, wetlands in excess of 1ha in size will be selected as larger wetlands are expected to perform better than smaller wetlands.

This study aimed to establish a methodology for wetland prioritisation on the basis of optimising the ecological services from wetlands with respect to faecal coliform pollution abatement of surface waters. The use of multi-criteria in a series of screening processes helped to indicate which wetlands in need of rehabilitation should be prioritised by the City of Tshwane. Study results must however be integrated into the broader integrated water resource management objectives of the municipality, so as to ensure that broader ecosystem processes regulating wetlands are protected. However, the study only made some recommendations for a site level assessment, and did not include site-specific investigations of the short listed wetlands.

Table 2.3: Size distribution of Wetlands (after United States Environmental Protection Agency, 2000, 15).

Size range in hectares	Cumulative percentage
Less than 1	46
Less than 10	75
Less than 100	93
Less than 1000	99

2.6 WATER QUALITY MODELLING

The modelling of waste input to surface water is broadly defined as point sources and non-point sources. Mass loading rate of each depends on the input flow and the input concentration of material (Venter et al., 1996). For linear systems such as rivers and streams, the concentration of a substance is due to multiple point source inputs or distributed sources (non-point sources). Substance concentration is consequently the sum of the responses due to the individual sources plus the response due to the upstream boundary condition of the river reach.

The quantification of mass rate of input of different water quality aspects and the comparison of these different variables of water quality has been incorporated by various authors in several rating methods, and include Multi Criteria Methods, Simple Additive Weighting technique and Simple Multi-attribute Rating Technique (SMART). A multi-criteria approach has been used by Greiner et al. (2005) for the assessment of relative impacts of non-point source pollution on the Great Barrier Reef. Scores were generated for each river basin against four criteria integrated into a single score for each basin, which allowed comparison of the river basins as well as the prioritisation of management alternatives. Values of criteria were obtained through stakeholder participation. The Simple Additive Weighting Technique has been used by Giupponi and Vladimirova (2006) to assess water quality. Weighted functions are used in the technique and are intended as an expression of judgement about how the ranges of variations of each indicator can contribute to the overall impact. Consequently, when using a GIS, values stored in the maps are made suitable for aggregation. SAW is then expressed by the following formula:

$IMPACT = I_i * W_i$

Where I = weighted indicator values stored in the map layers and

W = weights given to each indicator.

Total impact for each unit is then expressed by the following:

TOTAL IMPACT of Unit $I_{xy} = W_L * T_{xy} + W_H * H_{xy} + W_S * S_{XY}$, where T, H and S represent different aspects of vulnerability, drivers and pressures. Weights in the study have been equally distributed in every aggregation step, due to little evidence in the current literature regarding the relative importance of each indicator. The contributions of individual factors have been calculated as a percentage of the area that is being studied. The strength of the methodology and model, according to the authors, lies in its ability to identify areas of greater concern and allows policy makers to orientate themselves towards suitable intervention measures. The subjectivity of the weighting allocated to indicators, is however, a major source of inherent uncertainty in the modelling outcomes.

According to Pöyhönen and Hämäläinen (1997), simple multi-attribute value analysis is an overall value score that is attained for each alternative scenario. Alternatives are rated with respect to allocated weights. Weights are allocated through a 2-step process, including ranking the importance of changes in the attribute from best to worst, and ratio estimates are made on the relative importance of each attribute relative to the one ranked lowest in importance. Ten points are assigned to the attribute of least importance, and other alternatives are given points from ten upwards. Since SMART, a newer version has been introduced, where only the rank of attributes is used to derive weights. SMART has been used by Venter et al. (1996) to do a situational analysis of the microbial water quality in a peri-urban catchment in South Africa. Non-point source pollution has been assessed by using a simplified spatial model which considers run-off or pollution loading per unit area and down-stream attenuation, with drainage area as a scaling factor (Phillips, 1989). The element of downstream fluvial transport has been discounted by using a standard dilution model to enable estimations of the impact of run-off from a specific upstream area. Phillips (1989) concluded that this spatial framework is useful for quantitative risk-based comparisons. A great variety of techniques and models is to be found in the literature, and most are based on a criteria method, a grid method (such as the one used by

Phillips, 1989) or stochastic modelling of point sources. Most are fairly data intensive, or based on subjective rankings.

Current viewpoints with regard to modelling approaches are to match the modelling effort to the specific objectives of the study. Complex models are unnecessary when simpler ones will suffice (Mitchell, 2005). This is supported by Haydon (2006), who concluded that a relatively simple model can be used to predict pathogen levels with some degree of accuracy, whilst the more complex storm flow model used in the study appeared to be less accurate. Haydon (2006) further concluded that a limited amount of pathogen modelling coupled to hydrology has been undertaken, and that a comprehensive model for pathogen transport in catchments is still to be developed. A similar conclusion was reached by Elliot and Trowsdale (2007) who confirm that most of the models reviewed in their study on models for low impact urban stormwater drainage, have limited or no ability to predict pathogenic micro-organism or bacterial indicators, despite it being a major concern. In conclusion, little work has been undertaken on pathogen load estimations in storm water, and a methodology has been proposed in this study to start addressing this gap.

2.7 ASSIGNMENT OF WEIGHTS TO WATER QUALITY VARIABLES

Indicators vary greatly and are linked to the objectives of a project or program, such as using indicators of bacterial pollution of water sources for drinking water as opposed to other water uses (Venter et al., 1996). Indicators are also widely used to asses wetland condition, whether on a landscape level, or for intensive site assessments (Liu et al., 2006). Indicator links with specific programs or projects are further highlighted by Ehrenfeld (2000), who argues that appraisal should take place within a framework of social expectations, wetland capacities, and the need for active management and values unique to a particular urban context. As such, the use of multi-criteria assessments is required. Comparing various indictors with each other in order to rank aspects is accomplished by allocating relative weights to each indictor. For the purpose of this study, weighting will be done as far as possible, based on existing literature. Weights will be allocated to the different point and non-point sources assessed in this study.

Land-use has been highlighted in various studies as a risk factor for surface water quality (e.g. Pegram and Gorgens, 2001; Wright et al., 1993 and Murray et al., 2004). Land-use and water use ratings have been set at a 40:60 relationship by Murray et al. (2004) in an effort to quantify health risk associated with specific areas within a catchment. The relative contribution of point sources as opposed to non-point sources has been quoted as 20:80 (Wright et al., 1993). It must be mentioned that the study by Wright et al. (1993) was done in a typical South African city and is more relevant than the conclusion of Mitchell (2005) – who cites an 80:20 relationship based on studies in a developed country. The difference is due to the large number of settlements without adequate sanitation facilities, as well as failing sanitation systems (Wright et al., 1993; Jagals, 1995 and Van Vuuren, 2006).

Ratings between the different agricultural sources and their relative contribution to microbial water quality can be derived from studies of microbial content by the United States Environmental Protection Agency (2001) (Table 2.4). The use of figures provided in Table 2.4, is however, subject to the availability of data on the exact number of animals per feedlot or farm. An alternative would be to use information from Table 2.5, which is also provided by the United States Environmental Protection Agency (2001). Table 2.5 contains information on the typical cumulative run-off from an agricultural source as opposed to faecal coliform contribution by a single animal. It must be mentioned that the impact of run-off from any agricultural source on a water resource is determined by a variety of factors, such as distance from streams, the nature of the surface between the source and the stream, as well as management practices (Tian et al., 2002).

Faecal coliform loads from failing septic tank systems have been characterised by dividing the number of failing systems by the county area to obtain a density of failing systems per unit area expressed in acres (United States Environmental Protection Agency, 2001). The county density of failing systems was then multiplied by the area of each watershed to estimate the number of failing systems in each watershed. This number was used to calculate the estimated coliform loads from failing septic tank systems to the watershed, using an average water use, average number of persons and average coliform count in the excreta of each person (United States Environmental Protection Agency, 2001). The number of failing systems per area (expressed in acres) was calculated as follows: 205 failing systems/685,093 acres = 0.00046 failing systems per acre

This methodology has been adapted to express the annual reported sewage blockages in the Tshwane area for each river basin in the study area, as very little literature appears to be available on this particular aspect. This methodology was useful to estimate the total contribution to pollution from failing sewer lines in the study area. The flow rate and exact number of coliforms in effluent is not available, but the number of failing systems, total sewer line length per river basin and total river basin surface area have been used to express the density of failing systems as an expression of risk.

Effluent from municipal waste water treatment works is released directly into surface waters as opposed to other sources listed in Table 2.4. According to the information contained in Table 2.6, treated sewage effluent still contains significant numbers of faecal coliform bacteria, as well as TSS. The concentration of this pollution however, will vary in accordance with different waste water treatment techniques. Animal feedlot run-off is transported in rainwater run-off over various surface areas before reaching surface water bodies, presenting the opportunity for soil and vegetation attenuation of pollution. Surface areas impact on run-off quality in various ways, for example contaminants may be retained on soil grain surfaces or inside micro-interstices, or may percolate underground, depending on the physical characteristics of the soil (Nola et al., 2005).

According to the United States Environmental Protection Agency (2001), most pathogens are filtered out or attenuated in soil zones. Pathogens such as *E. coli* die within a matter of days (Pegram and Gorgens, 2001) and it is thus expected that the impact of effluent released directly into surface water bodies will be more substantial than pollution sources released some distance from a surface water body. It must be noted that literature varies on the extent of microbial die-off rates (Venter et al., 1997 and Tian et al., 2002). Raw sewage can also bypass waste water treatment works during extreme storm events, or due to human error or equipment malfunctions. Factors for the expected attenuation of microbial concentrations can be used to calculate relative contributions of waste water treatment works as opposed to other sources.

Table 2.4: Typical maximum concentrations in some pollution sources (after United States Environmental Protection Agency, 2001).

Pollution source	Typical concentration of faecal indicators
Pig	8.9 x10^9 organisms per day per animal
Cattle	100 x10^9 organisms per day per animal
Poultry	0.14 x10^9 organisms per day per animal
Dairy	100 x10^9 organisms per day per animal

Table 2.5: Summary of source specific faecal indicator concentrations (after United States Environmental Protection Agency, 2001).

Pollution source: point/area source type	Typical total faecal coliform values per 100ml
Animal Feedlot Run-off	1,25 x10^7 to 3,5 x 10^8
Raw sewage	2,3 x 10^7
Treated sewage effluent	10^4 - 10^6

Table 2.6: Typical maximum values of municipal waste water per 100ml (United States Environmental Protection Agency, 2001).

TSS influent maximum	TSS effluent maximum	Faecal coliforms (Influent maximum)	Faecal coliforms (effluent maximum)
587	39	10^7 to 10^9	10^4 to 10^6

2.8 SUMMARY AND CONCLUSION

Current viewpoints with regard to modelling approaches are to match the modelling effort to the specific objectives of the study. Complex models are unnecessary when simpler ones will suffice (Mitchell, 2005). It is also reiterated by Kotze et al. (2001) that wetland prioritisation should start with a set of strategic objectives. It is evident that there is no “blue-print” model to approach the problem of wetland rehabilitation in Tshwane.

Literature reviews on various studies indicate that point and non-point sources of pollution are seldom compared with each other in landscape level assessments, especially with reference to a single water quality parameter, such as microbial water quality. This is probably due to the fact that microbial water quality is a problem typically associated with developing countries, and the majority of studies are undertaken in developed countries. Where such studies are indeed undertaken, studies investigate surface water quality by means of surface water sampling and flow parameters. It is evident that a need exists for a water quality assessment technique which:

- Is suitable for a landscape level of assessment using spatially referenced data.
- Addresses the unique needs of developing countries as well as water resource managers at a local level.
- Enables the integration of the relative impacts of point and non-point sources on a landscape level without reverting to actual water sampling.

- Is cost effective and makes use of readily available local data and information.
- Is suitable for an initial screening and comparison of local catchments.

In relation to the utilisation of wetlands for water quality improvements, most studies emphasise the integration of a variety of wetland functions, of which the most prominent is biodiversity conservation. Within an urban context, many of these wetland functions have been lost, and protection for the sake of biodiversity outside of protected areas will only be achieved with considerable financial inputs. Sustainable wetland utilisation by local authorities and local communities is possibly the only avenue to ensure the protection and rehabilitation of wetlands considered to be unimportant from a representative habitat point of view. These wetlands still have an important role to play in providing ecosystem services on a macro level in urban areas, such as the improvement of water quality. As such, it is imperative to take a new perspective concerning the prioritisation of wetlands for rehabilitation in developed areas.

The value of wetlands depends on their hydrogeomorphic position in the landscape and the position of human settlements who find value in these ecosystems (Mitsch and Gosselink, 2000). Wetland area in proportion to the watershed or catchment is important. If wetlands are too small in relation, wetland functions will be overwhelmed and can no longer be realised (Mitsch and Gosselink, 2000). On average at least 5% of temperate-zone watersheds should consist of wetlands in order to optimise ecosystem services derived from wetlands (Mitsch and Gosselink, 2000). This figure will vary, however, depending on wetland type, catchment characteristics and water resource objectives. Wetland size, hydrogeomorphic position and relative position in relation to human settlements will be important criteria for wetland selection in this study. Selecting areas for wetland restoration in the past was largely determined by land ownership. A lack of a quantitative approach towards selecting and prioritizing, as well as a lack of a landscape perspective, implied that wetlands in the past were seldom considered for their contribution to ecosystem services on a catchment scale (Kotze et al., 2001). The large number of potential sites and the extensive site-specific fieldwork required makes large-scale regional restoration efforts difficult (O'Hara et al., 2000). Therefore, it is proposed that a screening methodology be used to prioritise catchments as a first step, with the selection criteria based on water resource problems and program goals. Due to the nature of water quality problems in the study area, it requires interventions to improve microbial water quality. However, a screening and prioritisation methodology to identify the basin most heavily polluted with pathogens is not readily available in the literature. An overview of the literature indicated that there are significant gaps in knowledge regarding the precise mechanisms of pathogen survival and mobilisation from the catchment store, and transport to surface waters. Areas requiring research include, amongst others, the assessment of catchment specific properties and features influencing the aforementioned aspects. The large number of variables involved in accurately predicting pathogen loads makes routine screening of catchments difficult and costly for water resource managers. A need therefore exists for a screening process suitable for local application in order to prioritize sub-catchments, before embarking on a site-specific investigation of wetlands in need of rehabilitation. Such a methodology is proposed in the following chapter.

CHAPTER 3

METHODODOLOGY

3.1 SELECTION OF METHODOLOGY

From the literature review, it becomes apparent that no “blue print” exists for the prioritisation of wetlands or their catchments, as the aim of the assessment guides the methodological approach. The aim of this assessment was to identify wetlands that could contribute to improved surface water quality after rehabilitation. The improvement of surface water quality is with specific reference to microbial quality. The identification of a specific wetland was based on a comparison of relevant pollution sources in each of the four river basins in the study area. Since the assessment is foreseen to be repeated in future for various catchments within the Tshwane area, the methodology had to address some inherent limitations, such as funding and data availability. The choice of methodology was also influenced by the fact that the study took place at a landscape level, as opposed to being executed at a site level. Given the large number of possible candidate wetlands, it was necessary to adopt a series of screening assessments. The methodology selected consequently had to be suitable for a landscape level of screening and assessment, be cost effective, utilise readily available data and integrate the relative impacts of point and non-point sources on microbial water quality.

A conceptual framework for the methodology used in the study is presented in Figure 3.1. The methodology proposed consists of 5 broad steps, as outlined. The description of the methodology refers to these steps, which have numbered for ease of reference.

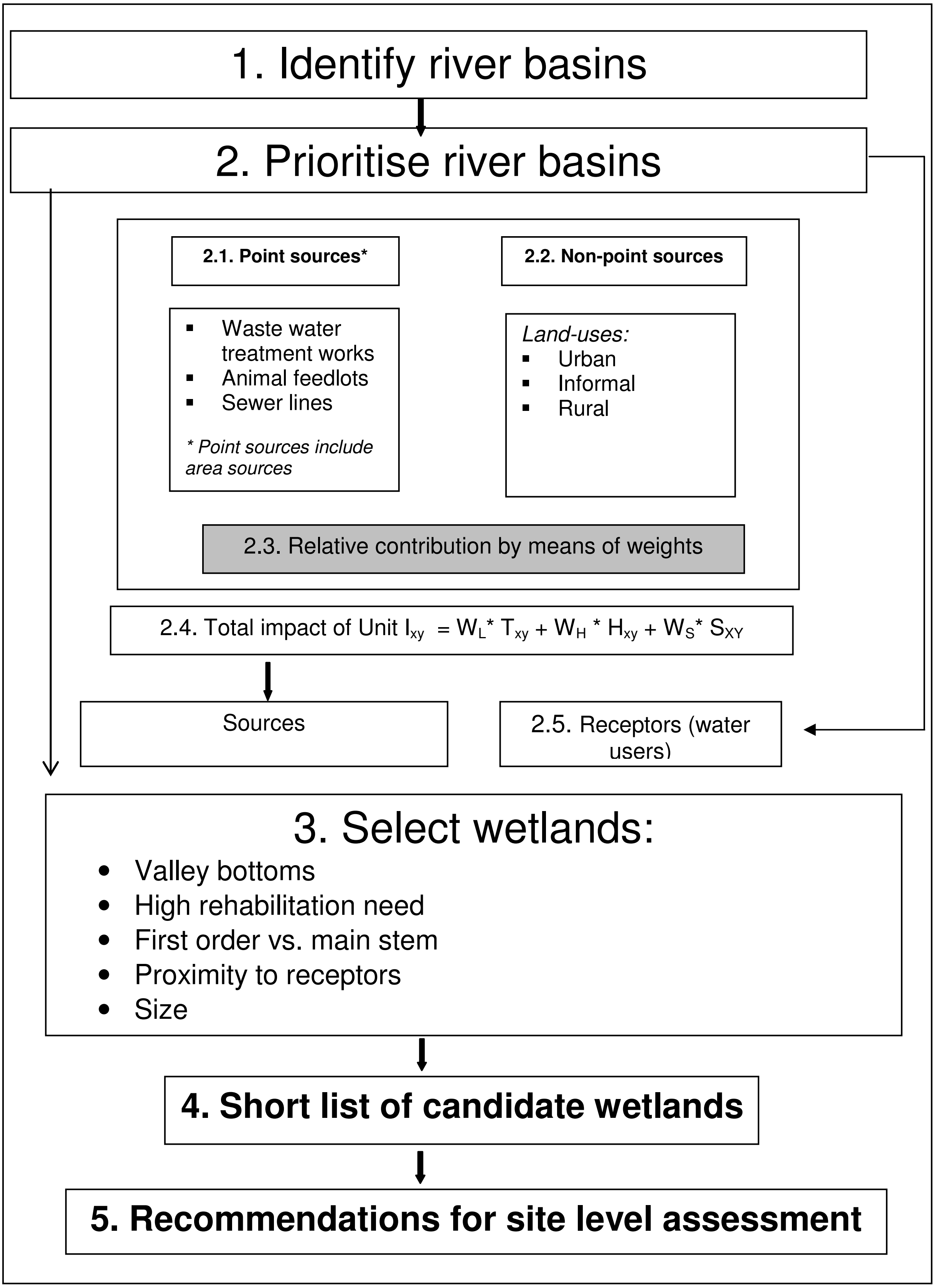

Figure 3.1: Conceptual framework of the methodology for the proposed study.

3.2 DATA AND SPATIAL DATA ANALYSIS

GIS is a powerful tool to analyse spatial distributions of a number of factors. A GIS (ARCView) was consequently used to generate composite layers of the following historical data in a series of screening processes:

- City of Tshwane wetland inventory data for the northern region of the city, comprising of a total of 192 wetlands.
- City of Tshwane land-use (cadastral) data.
- City of Tshwane basic services level data, as indicated by the status of settlements as either formal or informal.
- City of Tshwane basic services level (with reference to access to basic water and sanitation) data.
- City of Tshwane flood line data as compiled by the City of Tshwane Roads and Storm Water Division.
- Sewer line locations as compiled by the City of Tshwane Water and Sanitation Division.

The ability to integrate GIS with a variety of mathematical modelling further expands the utility of GIS for describing, understanding, and predicting spatial phenomena (Allan and Johnson, 1997). Mathematical modelling as a support tool to evaluate water quality remediation options is well documented according to Deksissa et al., 2004. Specifically the developments in water quality policy and strategies require mathematical models as a tool. The complex relationships between waste loads from different sources (point and non-point) and the resulting water quality responses of the receiving waters are best described with mathematical models (Deksissa et al., 2004). According to Zheng and Bennet (2002) the complexities regarding the fate, distribution and movement of contaminants require a multidisciplinary approach. The integration of GIS and numerical models has been used in a variety of studies (Brown and Joubert, 2003, Taebi and Droste, 2004, White et al., 2004 and Liu et al., 2006, amongst others). Therefore, it is evident that for the purposes of this study, a spatial analysis alone was not adequate. As such, GIS was used to generate composite layers of spatial data, but the relationship between data was expressed by means of an application of a numerical additive weighting technique. In order to prioritise the four river basins in the study area, the criteria on which the prioritisation is based had to be ranked. Ranking of these parameters was based on the attainment of a numerical score.

3.3 SAMPLING

The study area is located in the northern part of Tshwane as indicated in Figure 1.2. This includes the area north of the Magaliesberg range, which falls within the jurisdiction area of the Tshwane Metropolitan Municipality. The reason for the selection of the study area is based on the fact that at the time of the study, a wetland inventory for the northern part of the City only, was available. Four river basins, namely the Apies River, Sand River, Soutpan, and Stinkwater are situated in the study area. River basins in the study area were identified using existing spatially referenced data. The City of Tshwane Roads and Storm Water Division compiled the data to identify flood lines and assist with the compilation of a storm water master plan. The four river basins were consequently identified from this spatial layer. A part of the Apies River Basin falls out of the study area, but downstream water quality will be the result of activities within the upper reaches of the catchment thus all data related to the total catchment area is included in the study. The four river basins thus identified, represents the first step (number 1) as indicated in Figure 3.1.

As part of the prioritisation of the different river basins in the study area, point and non-point sources of faecal pollution were investigated. With reference to Figure 3.1, this section refers to numbers 2.1. and 2.2. Point and non-point source pollution that have been found in previous studies to contribute significantly to microbial water contamination were selected for inclusion in the study. Most authors focussed on different faecal pollution sources as part of their studies, but studies consistently point to the following as significant sources: urban land-use and population density factors (Pegram and Gorgens, 2001; Conway and Lathrop, 2005; Boyacioglu, 2006), waste water treatment works (Jagals, 1995; Murray et al., 2004; Gerba and Smith, 2005; Momba et al., 2006), sewer blockages (Wright et al., 1993; Van Vuuren, 2006), informal settlements without adequate sanitation (Jagals, 1995; Venter et al., 1997; Murray et al., 2004) and animal feedlots (United States Environmental Protection Agency, 1998; Pegram and Gorgens, 2001; Boyacioglu, 2006).

Relative weights, as calculated from information available in the literature and discussed in section 2.7, were allocated to the criteria for weighting of pollution sources during the prioritisation of river basins. Weights were used as the concentrations in and attenuation and die-off from faecal bacteria vary in different sources of faecal pollution. The use of weights allows for the relative contribution of each source type. Weights were based on findings of various authors. With reference to Figure 3.1 the allocation of weights represents step number 2.3.

The total projected impact of the different pollution sources on surface waters in each river basin was calculated using an additive weighting technique. After factoring in the relative contribution of each pollution source, the sum of these for each respective river basin was calculated. With reference to Figure 3.1, the number 2.4 represents the calculation of the total impact of various sources.

As indicated by Figure 3.1 (number 3), the receptors or users of poor quality river water were also considered. Water users that are vulnerable to infection due to a lack of access to treated water were selected based on their location within any of the four river basins. Information on the location of water users that are vulnerable due to lack of access to treated water was obtained from the City of Tshwane (2005b). These households include those making use of natural water sources, and/or unreticulated water points and/or have communal standpipes further than a 200m walking distance from their homes (City of Tshwane, 2005b). The terminology used by the City of Tshwane

(2005b) to describe these water users is “sensitive water users”. To avoid the repeated use of a long description for this type of water user in this document, the term sensitive users will be used. The result of steps one and two (Figure 3.1.) represented a first level of assessment or screening where a single river basin was selected based on the score achieved from the weighted criteria.

Observations and findings from various authors regarding wetland functionalities related to water quality improvement were used in the second phase of the study (Figure 3.1 number 3) for the selection of candidate wetlands. Candidate wetlands have been selected through a process of elimination from the total population list occurring within the selected river basin during the second level of assessment or screening. The total population of wetlands in this case, means those included in the wetland inventory, as well as those whose status has been indicated in the wetland inventory as wetlands with a high need for rehabilitation. The total population in this instance consisted of 43 wetlands, as indicated in Table 1.1.

3.4 PRIORITISATION OF RIVER BASINS

The prioritisation of the four river basins (step 2 as indicated by Figure 3.1) was based on an assessment of pollution sources of faecal coliform bacteria in each catchment. The aim of this part of the study was to identify the basin with the most pollution sources.

During the prioritisation phase, the following existing data were used:

- City of Tshwane cadastral data
- City of Tshwane Basic Services level data as indicated in the Strategic Plan for the eradication of Water and Sanitation Backlog in Tshwane (City of Tshwane, 2005b).
- The location and number of animal feedlots as supplied on request by the Environmental Health Practitioners from the City of Tshwane.

Data that represents the greatest possible risk for pollution and for sensitive water users were selected. For example, pathogen loads for different land-uses are indicated in the available literature within a range, and consequently the highest figure was selected. Rainfall data is available for the entire study area, which has been indicated by Venter et al. (2005) as averaging 619 mm per annum. The total stream flow for each river in the four river basins were expressed as the long-term rainfall average per annum for the study area divided by the total basin area. The annual contribution to stream flows from the waste water treatment works was then added to this figure. Run-off coefficients for the study area are not available; however, it was found that waste water treatment works contribute between 3 and 12% of stream flows based on rainfall when infiltration and evaporation have been excluded from the calculation (Table 3.1). The contribution of waste water treatment works to river channel flows is constant, and would constitute the main flow in channels during dry periods. As such, during these dry periods of the year persons utilising river water for household purposes would be utilising undiluted waste water effluent. Determining exact flows at any particular point in the study area, (taking into consideration run-off and infiltration in order to determine the exact dilution factor at that particular point), is not within the scope of the screening study due to a lack of available data. Given the risk-based nature of the study, where all factors were considered to contribute to the maximum extent, dilution would reduce risk accordingly. Factoring in stream flow dilution of pollutant load based on annual rainfall figures would mask the fact that at certain times of the year, flow in rivers containing waste water treatment works, consist of undiluted effluent, presenting a considerable risk to users.

Furthermore, with reference to pollutant load dilution, an assumption that larger catchments are likely to have larger flows and that bacterial pollution will be diluted cannot necessarily be made, as a linear relationship between these variables does not exist under all circumstances. The United States Geological Survey (2006) analysed pollution fluxes to some large rivers during a study conducted in 2003 and found that these fluxes were influenced by near-record river flows due to elevated precipitation. In contrast with expectations that larger flows would dilute pollution load, the elevated precipitation rather liberated large amounts of pollutants from the catchment store. High flow periods may have positive effects on water quality with dilution of surface waters by run-off, but on the other hand, run-off water can increase pollutant concentrations, thereby decreasing water quality (Boyacioglu, 2006). Dilution is based on the nature of the hydrological event such as rainfall intensity and duration, as well as frequency (Trask et al., 2004). Catchments can build up large stores of contaminants during dry periods, which are liberated during storm events and transported to rivers and streams via run-off. Provided that climate and geological characteristics are similar, larger catchments will have larger flows but anthropogenic factors such as water abstraction, effluent, land-use and the nature of water resource management in the catchment will ultimately dictate water quantity and quality. According to Mitchell (2005), diffuse urban loadings can vary by an order of magnitude or more between “clean” and dirty” urban catchments and that little effort has been made to identify sites that are significant in terms of load and potential on receiving waters. Consequently it was decided not to assume that larger catchments (specifically catchments situated in urban areas such as Tshwane) in the study area are likely to have larger flows, and that therefore bacterial pollution will be diluted.

The prioritisation of river basins took place in four phases. Weights and relative contributions were used between pollution sources within one group (as described in each of the phases in the following section), as well as between different pollution groups. Weights and relative contributions were also used between pollution sources and receptors (or users) of poor water quality.

In order to determine the river basin that should be prioritised for wetland rehabilitation, a simple additive weighting technique was used (refer to Figure 3.1, number 2.4). The total impact on a specific river basin was calculated as follows:

TOTAL IMPACT of Unit I_{xy} = W40 {W80* T_{xy} + W20 * H_{xy}} + W60 {S_{XY}}

Where:

Unit I	=	different river basins in the study area
W40	=	weight allocated to total pollution sources in a river basin
W80	=	weight factor allocated to land-use
T	=	specific impacts related to land-use
W20	=	weight factor allocated to diffuse and areal sources
H	=	specific impact related to diffuse and areal sources
W60	=	weight factor allocated to the number of users at risk of contaminated surface water
S	=	number of users at risk of contaminated surface water

The weighted values for pollution sources have been calculated by applying a weighting factor to the findings of previous sections. The values obtained provide an indication of the extent of the different pollution sources within each river basin.

The values representing the potential pollution loads originating from the different pollution sources in each river basin were added to obtain a single value. Relative weights were allocated for land-uses and point or areal sources in an 80:20 relationship (based on Wright et al., 1993). Final pollution load rankings are based on the sum of the land-use score, and the areal and point source score of each basin. This score was multiplied by a factor of 0.40, (to give effect to the 40:60 relationship between pollution sources and users at risk from contaminated surface water) effectively reducing the total pollution load weighting with 40%. The density of users at risk from contaminated surface water was multiplied by 0.60, and the sum of the two scores obtained, is used for the final ranking of the river basins.

Table 3.1: Relative contribution of waste water treatment works to annual volume of flow of rivers in the study area (m^3).

River basin	Flow	Effluent release from WWTW*	Total flow (litres)	Percentage contribution of WWTW* to total flow
Apies	492730	69474	562204	12.36
Sand	199937	1278	201215	0.63
Sout	109724	3504	113228	3.09
Stink	86660	0	86660	0.00

* Waste water treatment works

3.4.1 Compilation of scores for diffuse pollution

Four composite spatial layers (one for each river basin) have been compiled, based on river basin data and City of Tshwane cadastral data, in order to rate river basins based on quantities of diffuse sources of pollution related to general run-off. This step is indicated in Figure 3.1 as number 2.2. Data that was clipped onto these composite layers includes urban land-use, informal settlements, rural land-use, and sewer centre lines (used for calculations as explained in section 3.5.2). For the first phase, a GIS (ARCVIEW3) was utilized and the surface area of the total river basin, surface area of the various land-uses and the sewer line lengths per river basin area were calculated from the composite layers for each river basin. Weights were calculated for the different land-uses (maximum values have been used), based on studies by Pegram and Gorgens (2001) (Table 2.2). Utilising maximum values, the relative contribution of the different broad categories of land-uses could be calculated (Table 3.2). The contribution of each individual type of land-use was expressed as a percentage of the sum of the total maximum production rates of the three types of land-uses. Because the purpose of the study was not to calculate exact surface water bacterial loads, but express the magnitude of pollution sources relative to each other, only the contribution of each land-use type relative to each other was used as weights. For example, informal settlements can contribute up to 5 times (Table 2.2) the number of faecal bacteria than urban areas of equal size. The larger the surface area of informal settlements, the higher the faecal bacteria counts in surface run-off will be at a down-stream point. Consequently, basins with larger informal settlement areas will score higher points during the prioritisation phase.

Die-off ratios of pathogens for the study area are not available and have consequently not been taken into account. Minimum die-off rates would represent the maximum possible risk, and is therefore consistent with the study methodology.

Table 3.2: Relative weights allocated in the study to different land-uses.

Land-use Category	Areal production rates as per table 2.2	% Contribution in accordance with areal production rates	Relative Weight to be used in the river basin prioritisation phase
Informal	500	77%	0.77
Urban	100	15%	0.15
Rural	50	8%	0.08
Total	650	100	1

3.4.2 Compilation of scores for non-point and point source pollution

In this phase of the study (represented by number 2.6 as indicated by Figure 3.1), pollution point and non-point sources (other than related to specific land-uses) per river basin were calculated, using the composite layer generated in section 3.4.1. Although previous studies (e.g. Wright et al., 1993) indicate an 80:20 relationship between land-uses and point or discreet area sources (such as a site with intensive animal husbandry), the three sources mentioned above needed to be compared with each other, with reference to the relative contribution of each to the total pollution load. Equal weights could not be allocated, due to the die-off and attenuation impacts when polluted water flows over vegetated surfaces (originating for example from an animal feedlot), as opposed to effluent from a wastewater treatment works which is released directly into a stream. Blocked municipal sewage pipes normally drain into the nearest storm water conduit and main sewage pipes are installed in the lowest possible gradient point, implying their close proximity to natural watercourses. It is therefore evident that equal weights for the three respective sources in this category are not accurate. Weights are consequently allocated based on known values per volume of run-off or effluent, and secondly based on an attenuation factor in accordance with findings from Nola et al. (2006), regarding bacterial retention in soil.

The compilation of scores for point and non-point source pollution firstly included calculating the total effluent released into each river basin from the wastewater treatment works with information obtained from the City of Tshwane (2005b). Secondly, the number of animal feedlots per river basin were totalled, and expressed as a ratio of the total surface area of each river basin. The Environmental Health Practitioners of the City of Tshwane provided the location and number of animal feedlots in the study area. Factoring in specifically animal feedlots into the study were based on the realisation that a distinction based on land-use as per section 3.4.1 would not sufficiently express the risk associated with animal feedlots. Land-use zoning practices in the past generally considered undeveloped land as “agricultural”, irrespective of the actual activities on the land. Agricultural land consequently may or may not include activities pertaining to agronomy or animal husbandry. As such, not all agricultural areas can be considered as equal for the purposes of this study. Furthermore, the study area include previously disadvantaged areas with little economic development (Stinkwater and Soutpan River basins) as well as areas associated with more extensive economic activities (Apies River basin) and consequently more intensive use of land. The distribution of animal feedlots in the study area was then expected to vary greatly between the respective river basins.

A single figure relating to the annual number of blockages removed from municipal sewer systems was obtained for the study area as a whole from the City of Tshwane. Municipal sewer lines exclude connector lines to individual households and represent a “higher order” of sewer line. A “higher order” of sewer line is a line that service several households and is generally located in the street area or adjacent to water courses (the majority of sewer lines flow by means of gravity) as these occur at the lowest point in the landscape. The possibility of surface attenuation of bacterial load is consequently minimised. Data for this study related to the number of private sewer blockages was not available. The probability of surface-based attenuation of private sewer blockages is larger than for the higher order sewer lines. Due to the aforementioned, surface based attenuation for municipal sewer line blockages was not factored in. This is also in line with the worst-case scenario on which the study was based.

According to Mr Adriaan Odendaal (Personal communication, 2006) from the Water and Sanitation Division of the City of Tshwane, risk factors for blockages range from invasive tree roots to foreign objects and are not specifically related to any suburb or land-use within the Tshwane municipal area. It was therefore assumed that sewer lines within all the river basins have a similar probability of experiencing a blockage. For the purposes of this study, it was then assumed that sewer line length within a basin is proportional to the number of incidences of sewer blockages within a basin, such that the longer the sewer lines within a specific basin, the higher the probability for sewer blockages. A number of sewer blockages were allocated for each river basin based on its specific sewer line length relative to the sum of all sewer lines in the study area. This value was then used to determine the density of failing sewer lines per river basin, by dividing the number of blockages in each basin into the surface area of each basin. This was done to factor in basin size. As mentioned, sewage overflows resulting from municipal sewer lines are primarily carried directly to the storm water system by the water in these systems due to their location in streets with storm water infrastructure or being located next to water courses. Sewage overflows in this scenario is thus not greatly attenuated by soil or vegetation before reaching surface water bodies. Larger sewers are associated with the lowest possible gradient point in a landscape and are consequently associated with water courses. Most overflows from blockages will consequently flow directly into surface waters. Based on limited opportunity for surface attenuation of bacteria originating from sewer blockages, a higher ratio of impact has been allocated to sewer blockages when compared to run-off from animal feedlots. A potency figure, based on the most probable number of faecal coliforms per unit of pollution, has consequently influenced the final impact ratio. Based on this ratio, the different categories of point and non-point sources have been scored. Relative weights have been allocated in accordance with data from the United States Environmental Protection Agency (2001), who based data for each category on actual studies from various authors (Table 3.3). Information contained in Table 3.3 is presented as a range and the maximum values have been used for the purposes of this study. This is in keeping with the selection of the most extreme scenarios for variables in this study. The relative contribution of point sources is indicated in Table 3.4. Table 3.4 is based on the relative contribution of individual sources listed in Table 3.3 to the sum of the typical coliform values of these three sources.

Table 3.3: Summary of source specific faecal indicator concentrations (after United States Environmental Protection Agency, 2001).

Point/Area source type	Typical total faecal coliform values per 100ml
Animal feedlot run-off	1.25 x10^7 to 3,5 x 10^8
Raw sewage	2.3 x 10^7
Treated sewage effluent	10^4 - 10^6

Table 3.4: Relative weight attributed to pollution load as calculated for point sources, based on data from Table 3.3.

Point/area source type	Relative contribution of typical total faecal coliform values per 100ml
Animal feedlot run-off (represented by agricultural sources)	0.9381
Raw sewage (represented by sewage blockages)	0.0616
Treated sewage effluent (represented by waste water treatment works)	0.0003

The relative contribution of agricultural sources is the largest and treated sewage effluent contributes the least to faceal coliform values per 100ml of surface run-off or effluent.

Due to the fact that effluent from municipal wastewater treatment works and most sewage blockages are released directly into surface water bodies, surface-based attenuation of pathogen loads has been discounted. Purified sewage effluent still contains large numbers of faecal coliforms and according to the United States Environmental Protection Agency (2001), treated sewage effluent can contain 104 – 106 faecal coliforms per 100ml. It was therefore estimated that the relative contribution from waste water treatment works and blocked sewers would in effect be much higher than agricultural sources (Table 3.4). A conservative estimate in accordance with findings by Nola et al. (2006) was used for relative apportionment of pathogen loads from agricultural sources, waste water treatment works and blocked sewers. Bacterial retention potential of soils is influenced by a large number of variables such as physical, chemical, and structural properties of the soil layers. The ranges of total coliform detention in soil were calculated by Nola et al. (2006) as 69.22% to 99.95%. Using the most conservative approach (if 69% of all faecal coliform bacteria from agricultural sources is retained by soil), it was then estimated that the relative contribution of agricultural sources (Table 3.4) would be smaller than originally indicated. Consequently agricultural sources should, by relative contribution, only retain 31% of its original impact value. Given the other relative weights in the study (for example 60 receptors: 40 sources), the exact attenuation ability of local soil properties are not foreseen to influence the eventual outcome of the basin prioritisation phase in a significant manner. For the purposes of this study, weights have been allocated to sources within this section in accordance with Table 3.5. As can be seen in Table 3.5, the relative contribution of agricultural sources has been reduced to 31%. The relative contribution of point sources as opposed to land-uses as a diffuse source, has been quoted by Wright et al. (1993) as 20:80. Consequently, total pollution sources other than originating from urban run-off (as represented by land-uses) in this section, have been expressed against urban run-off (previous section) in a 20:80 relationship.

Table 3.5: Final relative contribution of typical faecal coliform values (per 100ml) used in the study of pollution sources other than urban-run-off.

Point/area source type	Relative contribution	Relative contribution accounting for surface attenuation	Final weighting factor used in the study*
Agricultural sources (represented by animal feedlots)	0.94	0.31	0.29
Raw sewage (represented by sewage blockages)	0.06	1	0.06
Treated sewage effluent (represented by waste water treatment works)	0.0003	1	0.0003

*Calculated as follows: column 2 multiplied by column 3.

3.4.3 Number of households at risk from contaminated surface water

Households at risk from contaminated surface water were calculated for each river basin, based on information supplied in the Strategic Plan for the Eradication of Water and Sanitation Backlog in Tshwane (City of Tshwane, 2005b). The determination of households at risk (or receptors of poor quality surface water) is indicated in Figure 3.1 as step number 2.5. Households at risk are those households that do not have direct access to piped drinking water in their homes, or access to a communal standpipe closer than 200 meters from their home (City of Tshwane, 2005b).

Careful consideration had to be given to factoring in households at risk. Larger catchments may have larger numbers of households at risk, but they may be distributed across and spread further apart than in smaller basins. In order to spread the benefit of improved surface water quality after wetland rehabilitation to as many households as possible, the statistical probability of a household enjoying this benefit will be greatest in the basins with the highest density of households at risk. For this reason the number of households at risk has been expressed as a density factor, and not purely as the number of households at risk from contaminated surface water. This is one aspect of the study though, that will benefit from a sensitivity analysis, as the spatial distribution of households at risk in a specific river basin will play a critical role in the final selection of a priority river basin. For this reason, the final selection of candidate wetlands for rehabilitation has to investigate the spatial distribution patterns of households at risk from contaminated surface water. Furthermore, the selection of more than one wetland for rehabilitation will increase the probability of bringing access to improved water quality to more households.

3.4.4 Final river basin ranking

The prioritisation phase was completed by comparing the total pollution load in a basin with the number of households at risk from contaminated surface water in a basin. Pollution load and water use by households at risk has been expressed as a 40:60 relationship (Murray et al., 2004) in an effort to quantify the health risk associated with specific areas within a catchment. In a similar way, this study has adopted a 40:60 ratio for pollution load to households at risk. Scores for all pollution sources and users at risk from contaminated surface water, were used for the final ranking of the river basins, utilising a Simple Additive Weighting technique (step 2.4. as indicated by Figure 3.1).

3.5 SELECTION OF CANDIDATE WETLANDS IN THE PRIORITY RIVER BASIN

A series of screening processes were used to select a short list of candidate wetlands (Figure 3.1 step 3 and 4). After each phase, data was clipped to the composite layer of the priority river basin. The query was based on a yes/no relationship and only wetlands compliant to the selection criteria were selected. The phases were as follows:

3.5.1 Selection of wetlands with a high need for rehabilitation

Wetlands with a low rehabilitation need were excluded from this selection phase, as Venter et al. (2005) indicated in the wetland inventory report to the City of Tshwane that wetlands with a low rehabilitation need were categorised as such, due to a possible threat to surrounding land-uses which might arise after rehabilitation. Projected alterations to the current hydrological functioning of the wetland could alter flood lines and may cause flooding of nearby houses. Wetlands with a low need for rehabilitation were eliminated based on the projected feasibility of rehabilitation, as found by Venter et al. (2005). Even after the elimination of this category of wetland, forty-three wetlands still remain, which had to be considered for prioritisation for rehabilitation. Given that funding is not available for the rehabilitation of all forty-three wetlands in the Tshwane area, the question of which wetlands to rehabilitate as a priority, still remained.

3.5.2 Selection of wetlands according to hydrogeomorphic location

Wetlands classified as valley bottoms and associated with a channel were selected due to the ability of these wetlands to improve surface water quality (Brinson, 1988; Dickens et al., 2003; United States Environmental Protection Agency, 2005).

3.5.3 Selection of wetlands according to minimum size

Wetlands larger than 1ha in size were selected as larger wetlands are considered more efficient for pollution attenuation than smaller ones, given the maximum possible retention time of water within the wetland in comparison to the total volumes of flow in the basin (Brinson, 1988; Suurballe, 1992; United States Environmental Protection Agency, 1993; Interstate Technology and Regulatory Council, 2003; United States Environmental Protection Agency, 2005; Kay et al., 2005). Size will be an important factor in the site level assessment, when areal removal rates have to be calculated.

3.5.4 Selection of wetlands according to location

Wetlands on the main river stem were selected, since effluent from all wastewater treatment works are released directly into the main river channels. Furthermore, all households at risk from contaminated surface water in the Apies River are located in the furthermost reaches of the basin, next to the main river channel. Due to their location, they will utilise water from the main river channel and are consequently at risk from all pollution sources located up-stream.

3.5.5 Proximity to receptors

Receptors are users of stream water, mainly for household purposes. Wetlands should be up-stream from and ideally as close as possible to households at risk from the utilisation of contaminated surface water. Wetlands also had to be located between these users and wastewater treatment works.

3.5.6 Storm water outlets

Ideally, storm water outlets, being a conduit for contaminated surface run-off, had to be located just upstream of wetlands in order for wetlands to have a positive impact on pollution loads. Storm water inflows, downstream from wetlands, will not be in direct contact with wetlands, and effluent quality will therefore not have the opportunity to be improved.

3.6 SUMMARY

The study was in essence an assessment of pollution sources and not a study of the actual pollution levels in surface waters. The study can be compared to a hazard assessment, and as such, maximum possible values were used. It was, however, of importance to replicate the same methodology and weightings across all four river basins to ensure consistency. Adapting the figures to account for a variety of possible scenarios is one way to estimate the consequent level of error inherent in the methodology used in this study. To this effect, sensitivity analysis could be a useful tool. Due to the many uncertainties associated with models attempting to predict water quality, Manache and Melching (2004) state that it is often necessary to perform a model reliability analysis prior to model application. Uncertainties are mainly due to the number of input parameters as well as their variability (Zheng and Keller, 2006). A sensitivity analysis of the Canadian water quality index undertaken by Gartner Lee Limited (2006) found that the index was less likely to incorrectly rank a site when more variables were included in the calculation. When the number of parameters was increased beyond ten, index scores improved significantly. It was also found that parameters based on at least three years of data further improved the reliability of the index. With reference to this study, sensitivity analysis can be useful specifically for the identification of parameters that can significantly influence model predictions, such as the impact of rain events on stream flow quality. This can determine whether additional parameters are required. The actual contribution of waste water treatment works to surface water bacterial loads is another area for investigation. Closer interrogation with the number of households in a basin as opposed to the density of households at risk from contaminated surface water is another critical matter for investigation. Sensitivity of the model predictions should also be analysed for different scenarios, such as flow volumes and attenuation rates. Based on a study by Campolongo et al., 2007, each input or variable used in this study can also be subjected to a number of incremental ratio's to determine whether the outcome of the study will change significantly. The accuracy of the risk estimations used in this study can also be determined with actual water sampling over an extended period of time.

CHAPTER 4

RESULTS

4.1 INTRODUCTION

This study aimed through a series of screening processes at a catchment scale to identify a single river basin as a first priority for wetland rehabilitation efforts. This study was based on the premise that naturally occurring wetlands in the study area could be rehabilitated to fulfil a water quality improvement function (Brinson, 1988; Hunt et al., 1996; Byström, 2000; Kay et al., 2005; Kieckbusch et al., 2006). Sources of faecal coliform pollution have been indicated as storm water originating from urban and agricultural run-off and waste water treatment works which typically release purified sewage water into river systems, but still contain large numbers of faecal bacteria. These pollution sources all contribute to deteriorating surface water quality. Several thousand households without access to piped drinking water, utilising surface water for drinking and other household purposes, compound the matter of providing a safe and healthy environment to communities.

This chapter will present the results of the study in accordance with the study framework, as presented in Figure 3.1, which provides an overview of the screening and prioritisation process that was followed. Study results are presented and explained. Study objectives and problem statements as indicated in Chapter One, are addressed. The chapter concludes with a summary of study results.

4.2 IDENTIFICATION OF RIVER BASINS

Four different river basins were identified in the study area, as depicted in Figure 4.1 and surface areas of each river basin are indicated in Table 4.1. These were calculated automatically by the GIS software as one of the standard tools of ARCView. A part of the Apies River Basin falls out of the study area, but downstream water quality will be the result of activities within the complete catchment. All data related to the complete catchment was included in the study.

The Apies River basin comprises more than half of the total study area (Table 4.1), whilst the Stinkwater basin is the smallest, comprising of less than 10% of the total study area. Demarcated boundaries of all river basins will enable the determination of land-uses and other activities within each of the respective basins. The calculated surface area of each basin will also enable the expression of relative densities of polluting activities.

4.3 PRIORITIZATION OF RIVER BASINS

Prioritisation of the river basins comprised the bulk of the analysis phase. The total projected pollution load in each river basin was compared with the number of households at risk from contaminated surface waters, within each river basin.

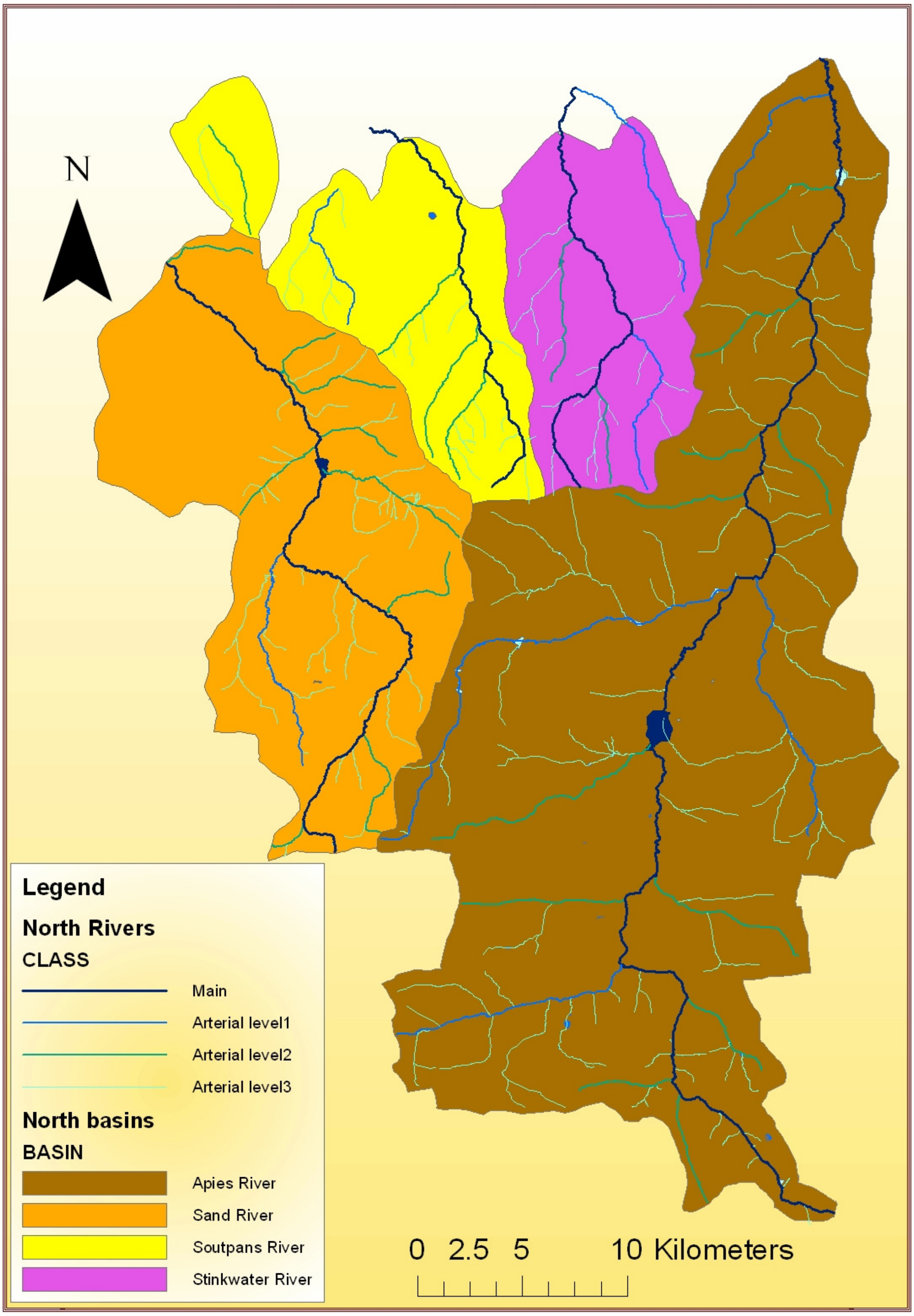

Figure 4.1: The four river basins contained in the study area.

4.3.1 The diffuse pollution loads in the river basins

River basins were rated based on quantities of diffuse sources of pollution related to surface run-off from the different land-uses. The respective surface areas of each of the three main land-use types were determined from the four composite spatial layers that were compiled (Table 4.1). The extent to which the Apies River basin is urbanised as well as the relative size of informal settlements in this basin in comparison to the other basins, can be seen. Weights (based on the maximum expected areal production rates of each land-use type in accordance with findings by Pegram and Gorgens, 2001) have been allocated to the surface area of the different land-uses as indicated in Table 4.2.

Table 4.1: The extent of land-use classes in the four river basins in the Tshwane area.

River basin	**Total basin area** (km^2)	**Urban**		**Rural**		**Informal**	
		km^2	%	km^2	%	km^2	%
Apies River	796	742.66	93	14.35	2	39	5
Sand River	323	140.13	43	179.27	56	3.6	1
Soutpan River	177	177	100	0	0	0.00	0
Stinkwater River	140	113.04	81	4.96	4	22	16

Table 4.2: Final ratings of land-uses in the four river basins in the Tshwane area.

River basin	**Urban**	**Relative weight 1**	**Rural**	**Relative weight 2**	**Informal**	**Relative weight 3**	**Final rating ***
Apies River	742.66	0.15	14.35	0.08	39	0.77	142.577
Sand River	140.13	0.15	179.27	0.08	3.6	0.77	38.1331
Soutpan River	177.26	0.15	0	0.08	0.00	0.77	26.589
Stinkwater River	113.04	0.15	4.96	0.08	22	0.77	34.2928

*Calculated as follows: River basin A: Final rating = (urban x relative weight 1) + (rural x relative weight 2) + (informal x relative weight 3).

The Apies River Basin is under severe pressure from a variety of pollution sources. Scores from each portion of this part of the screening study indicated that the Apies River Basin is likely to be the most heavily polluted by faecal bacteria. The Apies River basin consists of more than 93% urban area (Table 4.1), and drains a total of 742.66 km^2 of developed land, in comparison with the Soutpan basin, which is 100% developed, but only about a third the size of the Apies River Basin. The Apies River Basin by comparison contains an informal settlement area of almost 1.8 times the size of any of the other river basins. Each hectare of informal settlement contributes more than 5 times (Table 2.2) the most probable number of faecal coliform bacteria to a storm water system during a rain event, than any other land-use (with the exception of intensive animal farming). Various authors such as Wright et al. (1993), Jagals et al. (1995) and Pegram and Gorgens, 2001, have indicated that land-use is the overriding factor in surface water quality, and so the results of this part of the assessment are in accordance with expectations. The Sand River Basin is rated second, with a score of 38.13, but the score is significantly lower than the 142.57 value obtained for the Apies River Basin.

4.3.2 Analysis of pollution point and area sources per river basin

Three types of pollution point/area sources have been analysed, namely sewer line blockages, wastewater treatment works and intensive agricultural activities related to animal husbandry.

Sewer line lengths were established for each basin area using a GIS. The Apies River Basin, as expected, has the longest length of sewer lines due to its large urban area, which has been calculated at 2 400km. The Sand River Basin has the second longest sewer line length, which totals 688km. Blockages are based on the average total annual number of blockages removed by the City of Tshwane Metropolitan Municipality in municipal sewer lines (not connector lines to households) (Table 4.3.). All sewer lines were considered to stand an equal probability of experiencing a blockage, thus the higher the density of sewer lines within a basin, the higher the number of blockages within that specific river basin. Consequently, a proportion of the total number of sewer blockages was allocated to each basin depending on the length of sewer lines contained in a basin relative to the total length of sewer lines "available" in the whole study area. The number of blockages that can be expected to occur within a specific river basin was then divided into the specific basin surface area, to calculate the density of sewer blockages per river basin. Findings show that the Apies River Basin would experience the highest number of blockages, followed by the Sand and Soutpan River Basins (Table 4.4). The Apies River Basin will experience 41% of the total number of blockages in the study area, the Sand River basin 29%, and the Soutpan River 27%. The Stinkwater River basin contributes only 2.5% of total blockages in the study area.

Table 4.3: Projected average annual number of sewer blockages removed by the City of Tshwane Municipality.

Sewer blockages removed	Number
Year 2004/2005	20 900
Year 2005/2006	22 000
Projected Av. Pa.	21 450

Table 4.4: Projected number of sewer blockages per surface area in each river basin.

River basin	Sewer length (km) (1)	Relative length to total length in the study area (2)	Total nr of blockages in study area (3)	Number of projected blockages (4)	Basin Surface area (km^2) (5)	*Number of projected blockages per km^2 (6)
Apies River	2 400	0.692		14851.571	796	18.66
Sand River	688	0.199		4257.450	323	13.18
Soutpan River	353	0.102		2184.419	177	12.34
Stinkwater River	25	0.007		156.560	140	1.12
Total	3 466	1	21 450	21 450		

*Calculated as follows: River Basin A: [{(1)/3466)} x(21450)] / (5)

More than 93% of effluent released from waste water treatment works in the study area, are released into the Apies River (Table 4.5). This figure is nearly 6 times as much sewage effluent than the basin rated second for this factor. Due to the large relative contribution of waste water treatment works in comparison to the other pollution sources in the study area, combined with the high volume of effluent released into the Apies River Basin, it is evident that the Apies River Basin is under severe pollution pressure.

The Apies basin also contains more than 3 times the number of agricultural feedlots than any other basin (Table 4.6). The status of waste water treatment works, animal feedlots and sewer blockages have been calculated in Table 4.7. Agricultural feedlots and sewer blockages have been expressed as a density factor, to enable comparison of larger and smaller basins in terms of total surface area. The volume of sewage effluent is not expressed as a density factor, because effluent is released directly into river channels. In all instances, households at risk are located at the lower reaches of the river basins, implying that the full impact of surface water deterioration is felt at their particular location.

Table 4.5: Annual average discharge from municipal waste water treatment works.

River basin	Discharge (megalitres)
Apies River	6 9474
Sand River	1 277
Soutpan River	3 504
Stinkwater River	0
Total discharge released in study area	74 255

* Waste water treatment works

Table 4.6: Density of agricultural sources per river basin.

River basin	Number of pig farms	Number of cattle farms	Number of poultry farms	Total per river basin	Basin surface area (km^2)	Density of agricultural sources per km^2
Apies River	5	3	7	15	796	0.019
Sand River	2	1	1	4	323	0.012
Soutpan River	0	0	0	0	177	0
Stink River	2	0	2	4	140	0.029

Table 4.7: Summarised status of river basins with respect to point and area pollution sources.

River basin	Number of projected blockages per km^2	Density of agricultural sources per km^2	Annual average discharge from WWTW* (megalitres)
Apies River	18.66	0.02	6 9474
Sand River	13.18	0.01	1 277.5
Soutpan River	12.34	0	3 504
Stinkwater River	1.12	0.03	0

* Waste water treatment works

4.3.3 Households at risk from contaminated surface water.

Households at risk of contaminated surface water are expressed in terms of density. The Soutpan River Basin had the highest density of households without access to piped drinking water, followed by the Sand River Basin (Table 4.8). Although the Sand River Basin had the highest total number (6319) of this type of household, the density is slightly lower due to the larger size of the basin. The Stinkwater Basin has no users at risk from contaminated surface water located within its boundaries. The Apies River Basin has 2 704 households at risk from contaminated surface water, which is the second lowest score. In accordance with Murray et al. (2004), users at risk from contaminated surface water have been allocated a final rating of 0.60 as opposed to 0.40 for pollution levels. The total pollution load and the density of households at risk from contaminated surface water established in this part of the study have been utilised in the following section to complete the prioritisation exercise.

Table 4.8: Number of households at risk from contaminated surface water per river basin.

River basin	Number of households at risk from contaminated surface water	Density of households at risk from contaminated surface water
Apies River	2704	3.4
Sand River	6319	19.6
Soutpan River	3875	21.9
Stinkwater River	0	0
Total	12 898	

4.3.4 The Simple Additive Weighting technique

The final weighted scores for point and area source pollution has been calculated in Table 4.9 indicating that the Apies River basin contains the highest probable pollution load. The Sand River Basin has been rated second. It is evident that the Apies River Basin scored significantly higher values for this part of the analysis. This is largely due to the high volumes of sewage effluent. The simple additive weighting technique has been applied based on the values obtained in Table 4.9. The application of the simple additive weighting technique and final scores for each river basin in the study area are indicated in Table 4.10.

Final scores are illustrated in Figure 4.2, indicating that the Apies River Basin is ranked first, followed by the Sand, Sout and Stinkwater basins. The Apies River Basin has a significantly higher total score than any of the other basins (Table 4.10). Despite the Apies River Basin having a lower density of users at risk from contaminated surface water, the total pollution load in the basin will represent a far greater risk to users than surface water in any of the other basins. Figure 4.2 also illustrates the extent to which pollution sources and users at risk from contaminated surface waters contribute to the final score of each of the four river basins. The extent of pollution sources in the Apies River basin is well illustrated. The Stinkwater Basin cannot be considered for the identification of candidate wetlands, as no users at risk are located in this basin. Wetland rehabilitation efforts should thus be prioritised in the Apies River Basin in order to optimise wetland functionalities associated with water quality improvement. For the next phase of the study, only the Apies River Basin is investigated, with the focus to locate candidate wetlands for rehabilitation.

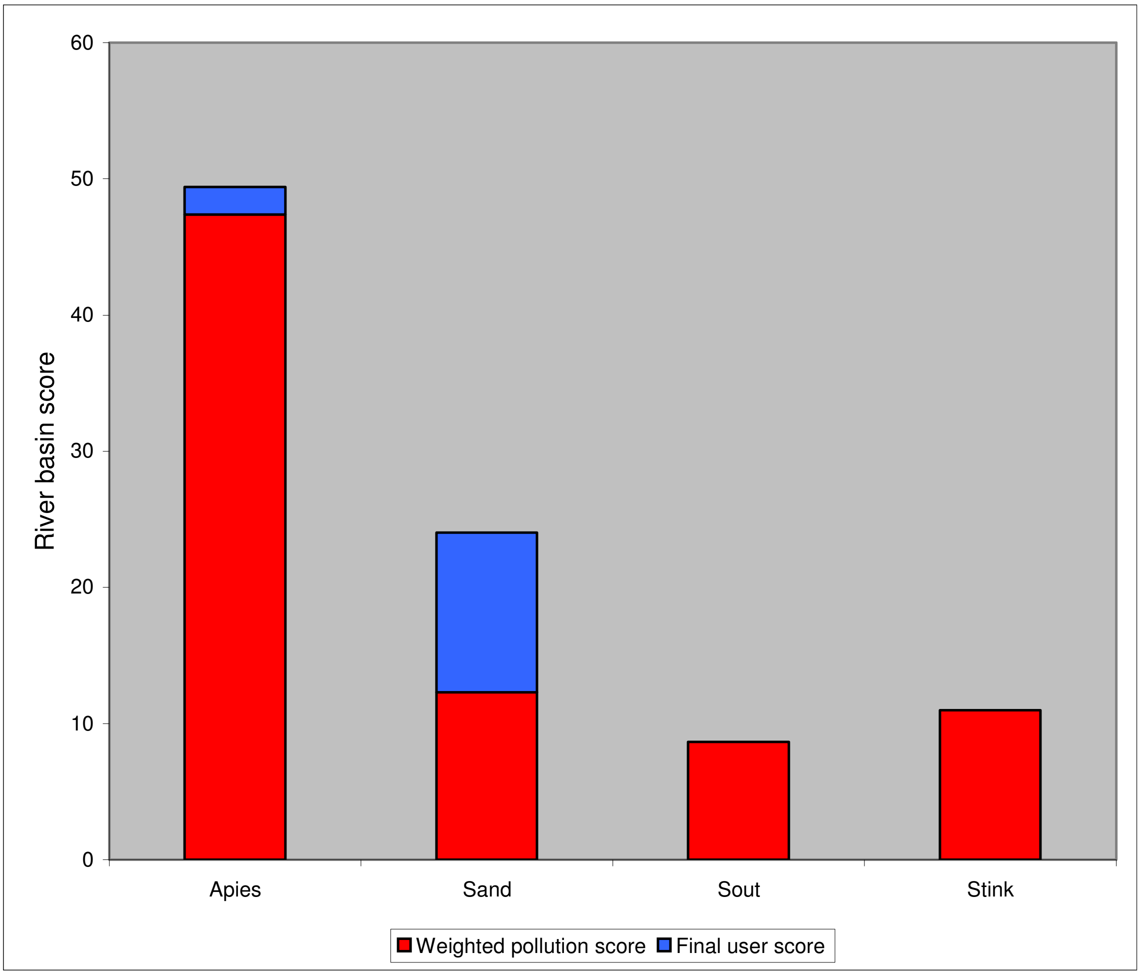

Figure 4.2: Relative contribution of pollution sources and users at risk to final basin scores.

Table 4.9: Final weighted scores per river basin for point and area source pollution.

	WWTW*	Weight 1	Total1	Blockages	Weight 2	Total2	Agriculture	Weight 3	Subtotal	Weight 4	Total3	Total
Apies	69 474	0.0003	20.85	18.66	0.0616	1.15	0.02	0.29	0.006	0.9381	0.006	22
Sand	1 277	0.0003	0.4	13.18	0.0616	0.8	0.01	0.29	0.004	0.9381	0.004	1.2
Sout	3 504	0.0003	1.1	12.34	0.0616	0.76	0	0.29	0	0.9381	0	1.9
Stink	0	0.0003	0	1.12	0.0616	0.07	0.03	0.29	0.009	0.9381	0.009	0.08

* Waste water treatment works

Table 4.10: Final Basin Scores, in accordance with the Simple Additive Weighting technique.

	Weight 1	Land-use score	Land-use subtotal	Weight 2	Areal and point source score	Areal and point source subtotal	Subtotal (all pollution sources)	Weight 3	Final weighted pollution score	Weight 4	Risk users per km^2	Final risk user score	Final basin ranking
Apies	0.80	142.58	114.06	0.20	22	4.40	118.46	0.40	47.38	0.60	3.40	2.04	49.42
Sand	0.80	38.13	30.51	0.20	1.2	0.24	30.75	0.40	12.30	0.60	19.56	11.74	24.03
Sout	0.80	26.59	21.27	0.20	1.9	0.36	21.63	0.40	8.65	0.60	0.00	0.00	8.65
Stink	0.80	34.29	27.43	0.20	0.08	0.02	27.45	0.40	10.98	0.60	0.00	0.00	10.98

4.4 SELECTION OF CANDIDATE WETLANDS IN THE PRIORITY RIVER BASIN

The second phase of the study investigated the total number of wetlands occurring in the study area within the Apies River Basin. Two relatively large wetlands (462 and 402 hectares respectively) in the upper reaches of the catchment complied to the criteria with respect to rehabilitation need, hydrogeomorphic location, size, and relative proximity to users at risk of contaminated surface water, as well as being directly associated with the river channel. Wetlands located in the upper catchment were not considered as candidate wetlands due to the distance between households at risk and wetlands. Wetlands had to be up-stream from households at risk, to be considered as candidate wetlands. Table 4.11 provides an overview of the main characteristics of the two wetlands identified after the application of the selection criteria. Attributes of all wetlands in the Apies River Basin are indicated in Annexure A. Wetland #1 is surrounded by informal settlements and has two waste water treatment works in its lower reaches. Wetland #2 is suitable for final selection due to its location in an agricultural area where no direct threat to property will result due to back flooding. Final selection of a particular wetland will depend on a number of factors such as the location of any waste water treatment works down stream of a wetland that will not enable the wetland to attenuate pollution load and the possibility of relocating the informal settlements currently occurring within Wetland #1. Wetlands #1 and 2 are in accordance with the selection criteria and their location have been indicated in Figure 4.3. All users at risk of contaminated surface water are located in the lower reaches of the basin, (as opposed to being spread across the basin area) which made selection of the candidate wetlands easier. Both of the selected wetlands are relatively large, exceeding 400ha. No storm water systems are associated with any of the two wetlands due to the nature of the land-use in the area. Waste water treatment works are located in the upper part of Wetland # 1, and wetland #2 is located after the waste water treatment works. Wetland #3 as well as most other wetlands in the focus area, are associated with small tributaries of the Apies River, and would thus be ineffective from a water quality improvement point of view, due to a lack of direct contact with surface water in the main river channel (Figure 4.3). Both wetland #1 (Figure 4.4) and 2 (Figure 4.5) are riverine wetlands and are elongate in nature. Other wetlands directly associated with the main river channel are consequently some distance away from where the users at risk from contaminated surface water are located. Due to this increasing distance, polluted surface run-off can re-enter surface waters. Due to the location of an informal settlement (which is a source of pollution but also contains households at risk from contaminated surface waters) in the immediate surroundings of Wetland #1 and the presence of a waste water treatment works in Wetland #2, it is advisable that both wetlands be earmarked for rehabilitation.

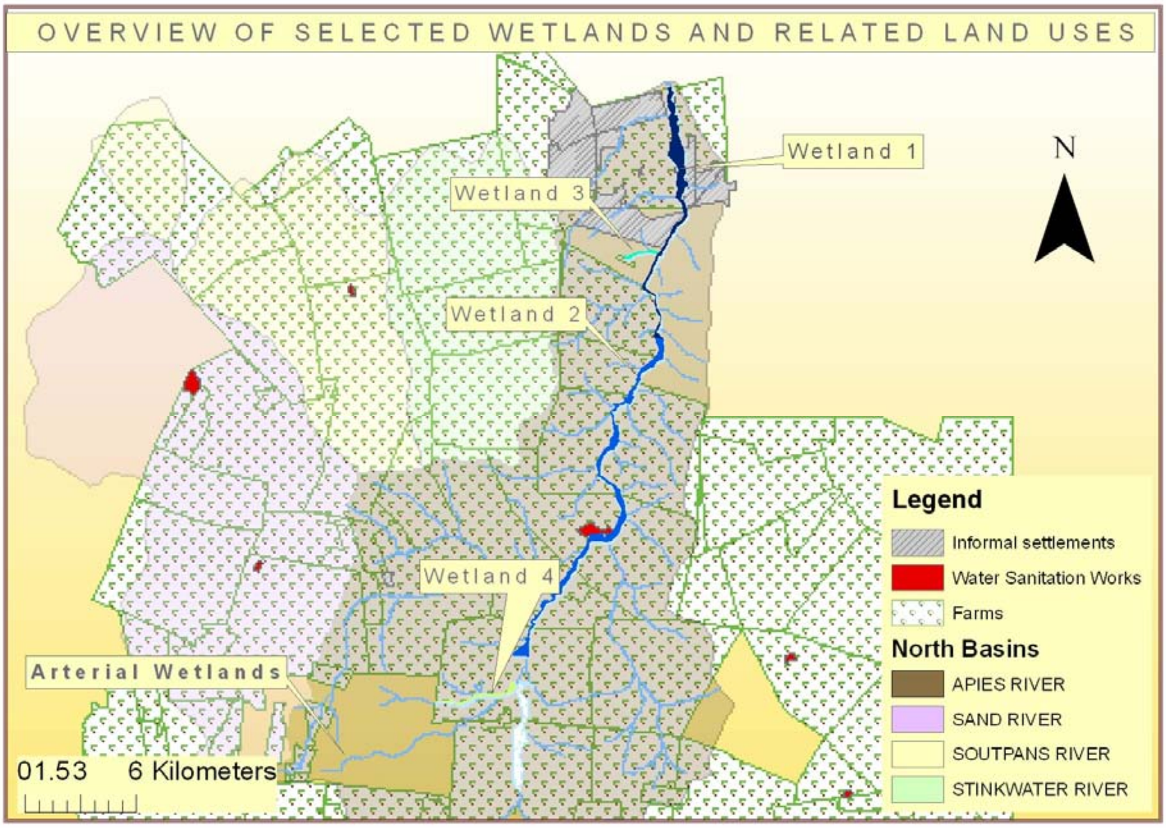

Figure 4.3: The Apies River Basin with an overview of related land-uses and candidate wetlands.

Table 4.11: Attributes of selected candidate wetlands.

Map number	River/ spruit	HGM* Type	Dominant vegetation	Rehabilitation need	Water regime	Size (ha)
2	Apies River	Valley bottom	Tall emergent vegetation	High	Permanent and seasonally wet soil	462.63
1	Apies River	Valley bottom	Tall emergent vegetation, grass, Acacia species	High	Permanent, seasonally and temporarily wet soil	402.09

*Hydro-geomorphic

Figure 4.4: Wetland # 1.

Figure 4.5: Wetland # 2.

4.5 SUMMARY

The study results indicate that the Apies River Basin has, in comparison to the other river basins, the most pollution sources contributing to poor microbiological water quality. Not only is the Apies River largely urbanised, but it contains several waste water treatment works contributing more than 12% to flow in the main river channel. It was also found that waste water treatment works are potentially the largest contributor to pathogen loads when compared to other pollution sources in the study area. Although the Apies River Basin did not have the highest density of users sensitive to contaminated surface water, the extent of pollution sources in the basin is such, that despite a 0.40 weighting allocated to pollution sources and 0.60 to sensitive users, it still obtained the highest score in calculations using the Simple Additive Weighting technique. Candidate wetlands were then selected for rehabilitation within the Apies River Basin. Two wetlands were selected; both are riverine wetlands associated with the main river channel and in relative close proximity to households at risk from contaminated surface water.

Chapter 5 will provide recommendations for site level assessments of candidate wetlands, as well as recommendations for the future application of the study methods used. Chapter 5 will also outline how this work can be expanded in a future study.

CHAPTER 5

CONCLUSION AND RECOMMENDATIONS

5.1 INTRODUCTION

The findings of Chapter 4 are firstly that the Apies River Basin has potentially the most sources of faecal coliform bacteria per surface area. The Apies River is under the highest pollution pressure, although it has a lower density of users sensitive to contaminated surface water. The extent of pollution sources in the basin is such that the risk for water users is particularly high. Secondly, the findings indicated two candidate wetlands for the purpose of rehabilitation. These two wetlands are ideally situated along the main system of the Apies River and located in the immediate vicinity of users at risk from contaminated surface water.

In this chapter, the results of the study will be discussed, recommendations made for site level assessments of the two candidate wetlands and the study methodology will be critically evaluated. It is also concluded that the study objectives have been met.

5.2 DISCUSSION

Current viewpoints with regard to modelling approaches in environmental science are to match the modelling effort to the specific objectives of the study. Complex models are unnecessary when simpler ones will suffice (Mitchell, 2005). It is evident from the study that no "blueprint" for surface water modelling exists, as the nature of the study area and the specific objectives will vary greatly between studies. This study made use of a Simple Additive Weighting technique to rank the respective river basins according to the extent of pollution sources and the density of users at risk from contaminated surface water quality. Determination of the relative weights of each pollution source type was the main challenge in the study. Since the study was limited to available data and was executed at a landscape level, the study methodology focussed on the relative density of pollution sources, as opposed to the determination of contaminated run-off volumes. This approach is particularly useful for a hazard-based assessment, as maximum values can be used, and pathogen die-off and surface attenuation can be largely excluded, except for the comparison of non-point and point sources.

The relative contribution of all pollution sources is based on values obtained from scientific literature. No dilution factor was considered in the study, given that the determinations of infiltration characteristics of the respective river basins are not available. However, it was determined that even if 100% of rainfall reaches surface waters, stream flow from three of the four river basins consists of large quantities of purified sewage effluent (between 3 and 12,3% of total annual flow volumes). During dry periods, flows will consequently consist mainly of purified sewage effluent in some of the river channels in the study area. The volume of sewage effluent has therefore been incorporated into the study. The risk-based nature of the study, in which worst-case projections have been considered, also allowed for the discounting of a dilution factor. Furthermore, the "first-flush" effect is a well known phenomenon where water quality parameters indicate significant increases of pollution concentrations (Larsen et al., 1998; Kim et al., 2003 and Prestes et al., 2006). Due to the extended residence time of viable microbes on the catchment store, it is expected that rivers in the study area will experience significant first-flush effects of faecal coliform bacteria due to the nature of the rainfall patterns in the study area.

The Apies River basin poses the greatest risk to users of surface water for household purposes. The catchment of the Apies River is largely urbanised and is 93% built up. Although ranked second, with reference to total surface area of informal settlements, the total surface area of informal settlements is almost double that of the Stinkwater basin, which has the second largest informal settlement area. Both these figures are very significant for the rating of pollution sources, as firstly, a study by Mallin et al., (2000) indicated that the most important anthropogenic factor associated with faecal coliform abundance in watersheds is percentage watershed impervious surface area. These surfaces, consisting of roofs, roads, driveways, parking lots etc., serve to concentrate and convey storm water borne pollution to surface waters. The study by Mallin et al. (2000) concluded that linear regression analysis indicated that percentage watershed-imperviousness alone could explain 95% of the variability in average faecal coliform abundance. Secondly, informal settlements are characterised by a lack of sanitation services and can contribute 500 times the number of faecal coliforms per surface area, when compared to other land-uses. Given the large informal settlements in the basin area and the urbanised nature of the basin, the ranking of the Apies River Basin as the most polluted during this phase of the study, is understandable.

The second largest contributor to faecal coliforms within the Apies River Basin is the large number of waste water treatment works, which contributes 54 times more total effluent than that of the Sand River Basin, ranked second. With regard to intensive animal husbandry or animal feedlots, the Apies River also contains the largest number of sources. It is only with respect to the number of users at risk of contaminated surface water, that other river basins scored higher. In particular, the Sand and Soutpan River basins contain large numbers of households without access to piped drinking water, but the extent of their risk is lower than similar users in the Apies River basin.

Given the fact that the Apies River Basin was ranked the highest priority during the Simple Additive Weighting technique, the screening process for candidate wetlands for rehabilitation was focussed on this particular basin. Two relatively large wetlands in the lower reaches of the study area complied with the initial criteria. Wetland area in proportion to the watershed or catchment is important. If wetlands are too small, wetland functions will be overwhelmed and can no longer be realised (Mitsch and Gosselink, 2000). According to Mitsch and Gosselink (2000), on average at least 5% of temperate-zone watersheds should consist of wetlands in order to optimise ecosystem services derived from wetlands. This figure varies amongst authors; for example Zedler and Kercher (2005) emphasised that pollution loads should determine the required wetland surface area. It is thus probable that both the candidate wetlands will be considered for rehabilitation purposes. The Apies River Basin has in total 864.26 ha of wetlands, implying that the basin, inclusive of all wetlands, consist of only 4.12% wetland area. The two candidate wetlands jointly make up only 1.09% of the basin surface area. Rehabilitation of both the candidate wetlands will have potentially significant impacts on down stream surface water quality.

5.3 RECOMMENDATIONS

For the final selection of a wetland earmarked for rehabilitation, a site level assessment of the two candidate wetlands must be executed. A set of weighted criteria should be compiled for all aspects of the investigation to enable a comparison to be made between the two wetlands. The length and cross section of the wetland must be determined, with reference to the ideal ratio of width and length. In this regard, a minimum width of 61m and minimum length of 15m has been recommended by the United States Environmental Protection Agency, 2000. Channel flow volumes, however, will be an important determining factor for optimum wetland dimensions. Various authors (Brown, 1985; Suurballe, 1992; United States Environmental Protection Agency, 1993) have concluded that hydraulic residence time in wetlands is of critical importance, thus extended residence times will improve water quality at the wetland outflow point. Relatively high water volumes in the Apies River channel are required to be in contact with the wetland, so as to enable reduced flow speed for enhanced residence times, as well as to enable other purification mechanisms, such as bacteriological predation to take effect. To increase residence time, vegetation type and vegetation density will be of importance, as surface roughness will enhance flow resistance and consequently hydraulic residence time. Given the fact that various authors alluded to the importance of wetland surface area relative to catchment surface area (Mitsch and Gosselink, 2000; Zedler and Kercher, 2005), it will be important to estimate the required wetland surface area relative to pollution loads in surface water in the Apies River. Other criteria to include will be the cost of rehabilitation. Conservation matters related to red data species occurring within any of the wetlands will make wetland protection an imperative, but has little relevance for water quality improvement. Conservation of valuable wetlands from a species conservation point of view should form part of a broader biodiversity conservation initiative. Site level assessments should also include an investigation into the opportunities for optimising ancillary benefits, such as biodiversity and measures to avoid undesirable impacts due to efforts to achieve a primary water quality improvement goal. Many wetland restoration projects are not successful, or only partially successful, because of a failure to recognise that wetlands form part of the larger landscape (Zedler, 1997). The long-term success of wetland rehabilitation will be determined by the extent to which anthropogenic disturbances can be limited, and it is therefore imperative that efforts to restore the wetland selected for rehabilitation, form part of a broader water resource and biodiversity management strategy. Small wetlands relative to their total catchment area, have reduced resilience to natural and man-made perturbations (Whigham, 1999), thus as many as possible of the wetlands within the Apies River basin should be earmarked for rehabilitation in a long term rehabilitation program.

5.4 CRITICAL EVALUATION

Most predictive models are based on particular assumptions about how landscapes and ecosystems function (Holl et al., 2003). However, quantifying these relationships accurately, will take years. It was therefore considered impractical to aim to quantify all relevant relationships and variables at a scale as large as the one represented in this study. This study has thus aimed to provide a basic methodology for prioritisation which is more viable for execution, given the limited amount of data available for the study area. The study therefore did not include a comparison of the findings with available water sample results. This is due to the limited nature and extent of water quality monitoring currently undertaken within the study area, which discounts many of the parameters required for such a comparison. For example, given the importance of the first-flush effect on water quality, mean event concentrations should be available to enable a comparative study, which however, is not available. Further research opportunities therefore include the comparison of the study results with improved water sampling results across the four different river basins, to determine the extent to which the study results and the actual surface water bacterial loads differ. However no proper receiving water quality management method can be defined solely on the basis of measured pollution concentrations in urban run-off and in waters receiving waste water treatment effluent because sanitary wastewaters are continuously discharged into receiving waters (Taebi and Droste, 2004). Surface run-off on the other hand, is discontinuous and transient. It has therefore been recommended that the unit load of contaminants may be a better basis for determining the management method of receiving water quality than measured pollution concentrations (Taebi and Droste, 2004). Assessments from a comparison study between sampling results, and the results from this study, should thus be interpreted against this recommendation.

The study did not take into account the impact of any septic tank or French drain systems on bacterial load, as this information was not available at the time of the study. For refinement purposes, it is recommended that future applications of this methodology also investigate the contribution of septic tank and French drain systems in the study area. Wetland rehabilitation initiatives seldom focus on the restoration of one functionality only, and given the relative scarcity of wetlands in the Tshwane area, the city must ensure the protection of as many wetlands and wetland functionalities as possible. Nevertheless, Grayson et al. (1999) concluded that in urban situations, wetlands are more likely to be restored to provide a specific function, such as the treatment of water, than to be restored to some apparent natural state. Restoration efforts often start with areas that are best preserved, an aspect which has not been included in the study. This is due to the specific management objectives of the prioritisation study which are specifically related to water quality improvement in the vicinity of users at risk from contaminated surface water quality. The study methodology will have to be adapted when users at risk from contaminated surface water are distributed across the catchment, as opposed to situated in a cluster in the lower reaches of a catchment, as is the case in this particular study. The study cannot be replicated in programs with different management goals, and is specifically designed for areas where microbial water quality is problematic. More studies are recommended to expand the list of candidate wetlands selected in this study, which should be based on a different set of water resource management goals. Finally, another perspective on the problem of wetland prioritisation could have been the selection of sites based on their proximity to pollution sources as opposed to users at risk from contaminated surface water.

Contributions that the study made to existing knowledge pertaining to water quality modelling and rehabilitation prioritisation include the development of a model which is suitable for South African water quality problems, which is essentially a rapid hazard identification based model with the aim of quantification of pollution sources and their potential contribution to bacterial load in surface waters. The integration of various pollution source types of bacterial load into a single model by allocating relative weights will add value to water quality management in South Africa, given that microbial water quality is of particular concern. Furthermore, the study contributed towards the development of a model suitable for screening purposes of catchments to enable wetland prioritisation and the provision of a new perspective on the value of wetland conservation and rehabilitation within a peri-urban South African context.

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Annexure A: Attributes of wetlands in the Apies River Basin

GPS/GIS metadata:
Projected Coordinate System: WG29
Projection: Transverse Mercator
False Easting: 0.00000000
False Northing: 0.00000000
Central Meridian: 29.00000000
Scale Factor: 1.00000000
Latitude of Origin: 0.00000000
Linear Unit: Meter (1.000000)
Geographic Coordinate System: GCS Hartebeesthoek 1994
Prime Meridian: 0

ID	Sub quaternary catchment	River name	HGM Classifi-cation	GPS latitude	GPS longitude	Dominant vegetation	Conservation status	Rehabilitation need	Water regime	Size (Ha)
59	A23F	Seepage.	S	-76802.93	-2817251	Grass and Acacia species	Vulnerable	High	Seasonally and temporarily wet soil.	17.96
60	A23F	Drainage line.	V	-78673.17	-2811787.5	Terrestrial vegetation	Problems	None	Temporarily wet soil.	111.49
61	A23F	Tributary of the Apies River.	V	-73758.83	-2805200.8	Grass and Acacia species	Problems	High	Seasonally and temporarily wet soil.	4.84
62	A23F	Tributary of the Apies River.	V	-75596.52	-2810693.4	Typha species Phragmites species sedge species terrestrial vegetation	Problems	Low	Permanent and seasonally wet soil.	88.53
64	A23E	Tributary of the Apies River.	S	-80746.1	-2822919.9	Grass species	Problems	None	Seasonally and temporarily wet soil.	3.28
65	A23E	Seepage.	S	-78521.3	-2823607.3	Grass and Acacia species	Good	None	Seasonally and temporarily wet soil.	2.85
66	A23E	Tributary of the Apies River (Grootvlei).	S	-74178.6	-2823696.9	Grass species	Problems	Low	Seasonally and temporarily wet soil.	2.38
67	A23E	Seepage.	S	-82755.6	-2825239.5	Grass species	Good	None	Seasonally and temporarily wet soil.	0.52
68	A23E	Tributary of the Metsi Metsuane stream.	S	-85375.4	-2826564.9	Grass species	Problems	Low	Seasonally and temporarily wet soil.	3.52
69	A23E	Tributary of the Apies River (Grootvlei).	S	-75256.6	-2827267.2	Grass species	Problems	Low	Seasonally and temporarily wet soil.	3.5
70	A23E	Seepage.	S	-77216.02	-2828128.6	Grass species	Good	None	Seasonally and temporarily wet soil.	7.7
71	A23E	Tributary of the Kaalplaasspruit.	V	-81695.1	-2829560.3	Tall emergent vegetation and grass species	Problems	Low	Seasonally and temporarily wet soil.	5.92
73	A23E	Tributary of the Kaalplaasspruit.	V	-88286.4	-2832243.7	Tall emergent vegetation	Problems	Low	Permanent and seasonally	1.08

ID	Sub quaternary catchment	River name	HGM Classifi-cation	GPS latitude	GPS longitude	Dominant vegetation	Conservation status	Rehabilitation need	Water regime	Size (Ha)
									wet soil.	
74	A23E	Tributary of the Apies (Makauvlei).	S	-79049.41	-2837672.3	Grass species	Problems	Low	Seasonal and temporarily wet soil.	1.62
75	A23E	Tributary of the Kaalplaasspruit.	S	-89099.9	-2833685.4	Grass species	Problems	Low	Seasonally and temporarily wet soil.	2.81
76	A23E	Tributary of the Apies River Boepensspruit	V	-83352	-2838099.7	Tall emergent vegetation	Problems	Low	Permanent & seasonally wet soil.	27.57
77	A23F	Tributary of the Apies River.	V	-75606.49	-2806351.6	Tall emergent vegetation and grass species	Problems	Low	Seasonally and temporarily wet soil.	92.5
78	A23E	Tributary of the Apies River.	V	-84142.2	-2837513.9	Tall emergent vegetation	Problems	High	Permanent and seasonally wet soil.	67.1
79	A23E	Tributary of the Apies River.	V	-77633.9	-2824982.6	Tall emergent vegetation and grass species	Problems	Low	Permanent and seasonally wet soil.	14.65
80	A23E	Drainage line of the Apies River (Grootvlei).	S	-73204.9	-2821113.6	Grass and Acacia species	Problems	None	Temporarily wet soil.	82.38
81	A23E	Tributary of the Apies River.	V	-76165.7	-2830825	Phragmites australis, grass and sedge species	Problems	Low	Permanent and seasonally wet soil.	104.7
82	A23E	Kaalplaasspruit (Tributary of the Apies River).	V	-82478.5	-2831087.2	Tall emergent vegetation	Problems	Low	Permanent and seasonally wet soil.	212.97
83	A23E	Metsi Metsuane (Tributary of the Kaalplaasspruit).	V	-87222.2	-2829533.6	Tall emergent and grass and sedge species	Problems	Low	Permanent and seasonally wet soil.	169.93
84	A23E	Tributary of the Metsi Metsuane stream.	V	-86108.8	-2827770.3	Tall emergent vegetation and grass and sedge species	Problems	Low	Permanent and seasonally wet soil.	119.04
85	A23E	Tributary of the Metsi Metsuane stream.	S	-85633.9	-2825169.6	Grass species	Problems	Low	Seasonally and temporarily wet soil.	14.84
86	A23E	Tributary of the Metsi Metsuane stream.	V	-85001.8	-2827584.2	Tall emergent vegetation and grass species	Problems	Low	Permanent and seasonally wet soil.	18.94
87	A23E	Tributary of the Kaalplaasspruit.	V	-81226.8	-2827910.9	Tall emergent vegetation and grass species	Problems	Low	Permanent and seasonally wet soil.	54.51
88	A23E	Tributary of the Apies River.	V	-78432.7	-2826137.5	Tall emergent vegetation, grass and sedge species	Problems	High	Permanent and seasonally wet soil.	124.31
89	A23E	Seepage.	S	-78997.4	-2823913.8	Grass species	Problems	Low	Seasonally and temporally wet soils.	12.53
90	A23E	Tributary of the Apies River.	V	-78202.8	-2823030.6	Tall emergent vegetation, grass and Acacia species	Problems	Low	Permanent and seasonally wet soils.	53.66
91	A23E	Triburary of the Apies River.	S	-75329.3	-2832179.9	Grass and sedge	Problems	Low	Seasonally and	131.02

ID	Sub quaternary catchment	River name	HGM Classifi-cation	GPS latitude	GPS longitude	Dominant vegetation	Conservation status	Rehabilitation need	Water regime	Size (Ha)
						species (Imperata cylindrica)			temporarily wet soil.	
92	A23E	Triburary of the Apies River.	V	-75424.5	-2828071	Tall emergent vegetation and grass species	Problems	Low	Permanent and seasonally wet soil.	24.72
93	A23E	Tributary of the Apies River (Grootvlei).	S	-73624.5	-2824583.4	Grass species	Problems	Low	Seasonally and temporarily wet soil.	26.38
94	A23E	Tributary of the Apies River (Grootvlei).	V	-74845.4	-2823291.5	Grass species	Problems	Low	Seasonally and temporarily wet soils.	3.58
95	A23E	Drainage line of the Apies River (Grootvlei).	V	-73676.3	-2822756.5	Grass and Acacia species	Problems	None	Seasonally and Temporarily wet soil.	27.05
96	A23E	Apies River	V	-75661.3	-2821242.5	Tall emergent vegetation	Problems	High	Permanent and seasonally wet soil.	462.63
98	A23E	Tributary of the Apies River	V	-81831.2	-2825618.9	Tall emergent vegetation, grass & Acacia species	Problems	Low	Permanent and seasonally wet soil.	32.43
99	A23E	Tributary of the Apies River	V	-81497.2	-2834310.2	Grass and trees.	Problems	Low	Permanent and seasonally wet soil.	11.57
100	A23F	Tributary of the Apies River	V	-76921.36	-2816264.6	Tall emergent vegetation, grass and Acacia species	Vulnerable	High	Permanent and seasonally wet soil.	74.1
101	A23F	Tributary of the Apies River	V	-74981.54	-2813411.9	Tall emergent vegetation, grass and Acacia species	Vulnerable	High	Seasonally and temporarily wet soil.	19.64
102	A23F	Apies River	V	-75241.7	-2818346.2	Tall emergent vegetation, grass and Acacia species	Problems	High	Permanent, seasonally and temporarily wet soil.	402.09
112	A23E	Apies River	V	-81677.6	-2839674.9	Tall emergent vegetation (Phragmites australis).	Threatened	Low	Permanent and seasonally wet soil.	279.32
113	A23E	Tributary of the Apies River (Makauvlei)	S	-80153.2	-2837696.6	Grass species	Problems	None	Seasonal and temporarily wet soil.	4.21
114	A23E	Tributary of the Apies River (Makauvlei)	V	-80630.9	-2837145.8	Down stream Typha capensis. Imperata cylindrica.	Problems	Low	Seasonal and temporarily wet soil.	1.36
115	A23E	Seepage (Grootvlei)	S	-79057.5	-2837676.1	Grass species	Problems	High	Seasonally and temporarily wt soil.	19.83
116	A23E	Tributary of the Apies River (Grootvlei)	S	-75655.34	-2824747.2	Grass species	Problems	Low	Seasonally and temporarily wet soil.	7.36
117	A23E	Tributary of the Apies River (Grootvlei)	S	-76415.31	-2824345.1	Grass species	Problems	Low	Seasonally and temporarily wet soil.	1.78
118	A23E	Tributary of the Apies River	V	-73426.4	-2819372	Grass species	Problems	Low	Seasonally and temporarily	5

ID	Sub quaternary catchment	River name	HGM Classifi-cation	GPS latitude	GPS longitude	Dominant vegetation	Conservation status	Rehabilitation need	Water regime	Size (Ha)
									wet soil.	
119	A23E	Drainage line of the Apies River	V	-78486.8	-2821192.2	Grass and Acacia species	Problems	Low	Seasonally and temporarily wet soil.	31.19
120	A23E	Seepage	S	-78486.8	-2822576.4	Grass species	Problems	High	Seasonally and temporarily wet soil.	1.48
121	A23F	Seepage	S	-74865.31	-2817614.7	Grass species	Problems	Low	Seasonally and temporarily wet soil.	7.26
122	A23F	Tributary of the Apies River	V	-72913.36	-2818376.6	Grass and Acacia species	Vulnerable	High	Permanent and seasonally wet soil.	44.76
123	A23F	Tributary of the Apies River	V	-72977.94	-2816936.1	Grass and Acacia species	Problems	Low	Seasonally and temporarily wet soils.	37.08
124	A23F	Seepage	S	-73197.06	-2815555.3	Grass and Acacia species	Problems	Low	Seasonally and temporarily wet soil.	7.85
125	A23F	Seepage	S	-73415.12	-2815335.4	Grass and Acacia species	Problems	Low	Temporarily wet soil.	2.28
126	A23F	Seepage	S	-74117.05	-2814165.2	Grass and Acacia species	Problems	Low	Temporarily wet soil.	4.12
127	A23F	Seepage	S	-72486.13	-2813090.1	Grass and Acacia species	Problems	Low	Seasonally and temporarily wet soils.	2.49
128	A23F	Seepage of the Apies River	S	-72101.98	-2810812.3	Grass species	Threatened	None	Seasonally and temporarily wet soil.	52.17
131	A23F	Seepage	S	-78339.97	-2815975.4	Grass and Acacia species	Vulnerable	High	Seasonally and temporarily wet soil.	80.2
132	A23F	Tributary of the Apies River	V	-76244.98	-2814629.3	Grass and Acacia species	Problems	Low	Seasonally and temporarily wet soil.	22.92
133	A23F	Tributary of the Apies River	V	-74895.81	-2812745.4	Tall emergent vegetation, grass and Acacia species	Problems	High	Seasonally and temporarily wet soil.	14.94
134	A23F	Seepage	S	-74097.33	-2804755.2	Grass species	Problems	None	Seasonally and temporarily wet soil.	1.33
145	A23F	Seepage	S	-80177.66	-2819506.8	Grass and Acacia species	Problems	None	Seasonally and temporarily wet soil.	8.88
157	A23E	Tributary of the Apies River (Grootvlei)	S	-75026.15	-2824008.9	Grass species	Problems	Low	Temporarily wet soil.	2.65