An integrated planning approach for the conservation of freshwater ecosystems in South Africa

by

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DECLARATION

I the undersigned declare the work presented in this thesis is my own original work, and has not been presented in part or wholly for the award of a degree in any university. However, Chapter 5 is in press in Biological Conservation, with my supervisors and another person who provided raw data as co-authors. The rest of the Chapters are in various stages of preparation for submission to international and local journals for publication.

Signature *Date*

Name in full

Mao Angua Amis

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ABSTRACT

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Freshwater ecosystems underpin the fabric of society and the environment, providing essential ecosystem services such as water and food, upon which all human beings depend. In order to secure these vital services requires the sustainable management of freshwater ecosystems. At present however, freshwater biodiversity is under severe threat from anthropogenic disturbances, and the situation is expected to worsen due to population growth and global change. Coupled with the threats to freshwater biota are the limited resources available to secure their protection. There is a need to therefore prioritise freshwater ecosystems in a comprehensive, adequate and representative manner to maximise the outcomes of conservation effort.

The concept of systematic conservation planning was developed to address this challenge. It offers a suitable framework for achieving conservation goals in the face of other competing land uses. The principles of systematic conservation planning are being widely applied in terrestrial and marine ecosystems, but their application in freshwater ecosystems is still relatively limited. Freshwater ecosystems provide challenges to conservation planning that are unique from those of terrestrial and marine ecosystems, such as the longitudinal nature of river systems and the associated connectivity, and catchment divides that constrain some obligate species. As a result freshwater ecosystems require conservation planning tools and approaches that are specifically geared towards addressing these unique challenges. Progress has recently been achieved in addressing some of these challenges, but there are still other outstanding issues that have not been comprehensively addressed. The aim of this thesis was to develop new frameworks, and test approaches for the application of systematic conservation planning principles in the conservation of freshwater ecosystems in South Africa.

The thesis addressed a range of issues along the systematic conservation planning continuum from biodiversity assessment to implementation. I first carried out a focused review of

systematic conservation literature between 1987- 2006, to gauge the extent to which freshwater ecosystems have being integrated in conservation assessments. Most of the focus was found to be on terrestrial ecosystems with minimal incorporation of freshwater biodiversity. Wetlands for example, were in most cases incorporated into conservation assessments without taking their diversity into account. This was partly attributed to the difficulty of classifying wetlands. I therefore developed and tested a hierarchical GIS framework for automating wetland classification as a strategy for incorporating wetland biodiversity, functions and benefits into broad scale conservation planning.

Based on the gaps that were identified in freshwater conservation planning and the approaches developed, I used the Cape Floristic Region, which is a global biodiversity hotspot to demonstrate how to effectively plan for freshwater biodiversity persistence. This was achieved through a comprehensive systematic conservation plan that incorporated the different freshwater ecosystems, assessed their ecological integrity and the extent to which freshwater and terrestrial biodiversity priorities overlap. It was found that freshwater and terrestrial priority areas are poorly aligned, with terrestrial priority areas grossly inadequate for achieving freshwater biodiversity targets.

In order to improve the poor alignment of freshwater and terrestrial priority areas, a GIS based protocol was proposed for integrating freshwater and terrestrial priorities in planning for biodiversity conservation in Mpumalanga. Following extensive scenario analysis in GIS and a widely used decision support system (MARXAN), it was found that goals for freshwater and terrestrial biodiversity could be adequately achieved when freshwater priority areas are used to drive the selection of terrestrial priorities in conservation planning, without compromising their unique characteristics.

In cognisance of the fact that the ultimate goal of systematic conservation planning is to ensure the implementation of effective biodiversity conservation strategies, a comprehensive assessment was undertaken to evaluate the management effectiveness of key agencies responsible for the implementation of freshwater conservation strategies in South Africa. The assessment focused on the strengths and weaknesses of current mechanism in the protection of freshwater ecosystems, with the aim of understanding how conservation planning products can be developed to specifically address the implementation challenges that arise. It was

found that a good regulatory framework, shared values, good learning basis, and adequate monitoring and communication were some of the main strengths of current mechanisms required for effective implementation. Meanwhile major constraints to implementation include inadequate capacity, misaligned strategies and inadequate alignment of monitoring with conservation objectives.

Keywords: *freshwater biodiversity, wetlands, systematic conservation planning, integration, management effectiveness, implementation*

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CHAPTER 1.INTRODUCTION

1.1 The status of freshwater ecosystems

Freshwater ecosystems which comprise all inland aquatic systems including rivers, wetlands, lakes, estuaries and groundwater are vital for the sustenance of human welfare (Naiman & Turner 2000, Higgins 2003, Fitzsimons & Robertson 2005). They are critical to all aspects of society, where they provide important ecosystem services such as food, water, energy, transport, and waste assimilation. Freshwater ecosystems are however, some of the most degraded ecosystems compared to terrestrial and marine ones (Ricciardi et al. 1999, Abell 2002, Saunders 2002). This has resulted in a freshwater biodiversity crisis, with more than 30% of freshwater vertebrate species extinct (IUCN 2008). The freshwater biodiversity crisis is partly attributed to global environmental change; the principle agents of which are; land use, anthropogenic changes in biogeochemistry, and biotic additions and losses (Lake et al. 2000). Flow modification, water pollution, habitat degradation, and invasive alien species have led to irreversible changes of freshwater ecosystem processes and functions in many regions of the world (Jansson et al. 2000, Dudgeon et al. 2006).

In Southern Africa there is severe water stress (Arnell 2004), and the situation is expected to worsen due to climate change (van Dam et al. 2002, IPCC 1998), population growth and changes in consumption patterns (Scholes & Biggs 2004). Unlike their terrestrial counterparts, freshwater biodiversity are often not legally protected through a system of reserves or national parks. Even some of the freshwater systems found inside protected areas are subject to land use impacts (Roux et al. 2008). In a recent assessment, 84% of South Africa's main stem rivers were classified as threatened (Nel et al. 2007). In the Kruger National Park for example, the headwaters of the main rivers flowing through the protected area are located outside the park, thus failing to protect these freshwater systems from the effect of upstream land use and other anthropogenic disturbances (Roux et al. 2008). Similar cases of poor coverage of freshwater biodiversity in terrestrial protected areas have been reported in other parts of the world (e.g. Barret & Ansell 2003, Mancini 2005). The poor protection of freshwater ecosystems is attributed to ad hoc reservation measures (Margules & pressey 2000, Ferrier 2000), where protected area delineation was mostly informed by terrestrial biodiversity without the consideration of freshwater biodiversity (Keith 2000, Filipe

2004, Linke et al. 2008). The inadequate protection accorded to freshwater biodiversity has led to calls for the establishment of conservation strategies to address the uniqueness of freshwater ecosystems, so as to offer better protection to freshwater biodiversity (Saunders et al. 2002, Abell et al. 2002, Kingsford & Nevill 2005).

1.2 Systematic conservation planning,

Systematic conservation planning offers robust tools for developing effective conservation strategies critical for addressing the challenges to freshwater conservation. Systematic conservation planning was first pioneered in terrestrial ecosystems in response to the *ad hoc* approach to delineating nature reserves. For a long time the delineation of protected areas was not driven by objective conservation principles and as a result most nature reserves were located in areas of limited land use opportunities in order to minimise conflicts between humans and conservation (Pressey 1994). Conservation planning was therefore devised to change this trend so that the location of important areas for biodiversity conservation are identified using systematic and scientifically defensible principles (Margules & Pressey 2000). The basic principles of systematic conservation planning comprise: representativeness, irreplaceability, complementarity, efficiency, adequacy and flexibility (see Margules & Pressey 2000 for more details). These principles are applied at various stages of the conservation assessment process (Fig. 1.1).

Systematic conservation planning is a continuum from biodiversity assessments to implementation (Knight et al. 2006). Biodiversity assessments involve the identification of spatial biodiversity priorities through a rigorous process of data collation on critical biota, status assessments and the assigning of biodiversity value to places (Margules & Pressey 2000, Groves et al. 2002, Higgins et al. 2005, Moilanen et al. 2007). Biodiversity assessments also address issues of integrating different ecosystems and biotas when identifying priority areas to ensure efficiency and the general robustness of the spatial biodiversity plan. The other end of the spectrum in the systematic conservation planning continuum deals with the development of strategies for the implementation of the spatial biodiversity priorities identified during the assessment phase (Knight et al. 2006). The implementation strategies determine the specific conservation actions that are required to be undertaken on the ground. Unfortunately most conservation planning efforts focus largely on biodiversity assessments, which failed to be implemented, often referred to by some authors as an implementation crisis (Salafsky et al. 2002, Knight et al. 2006).

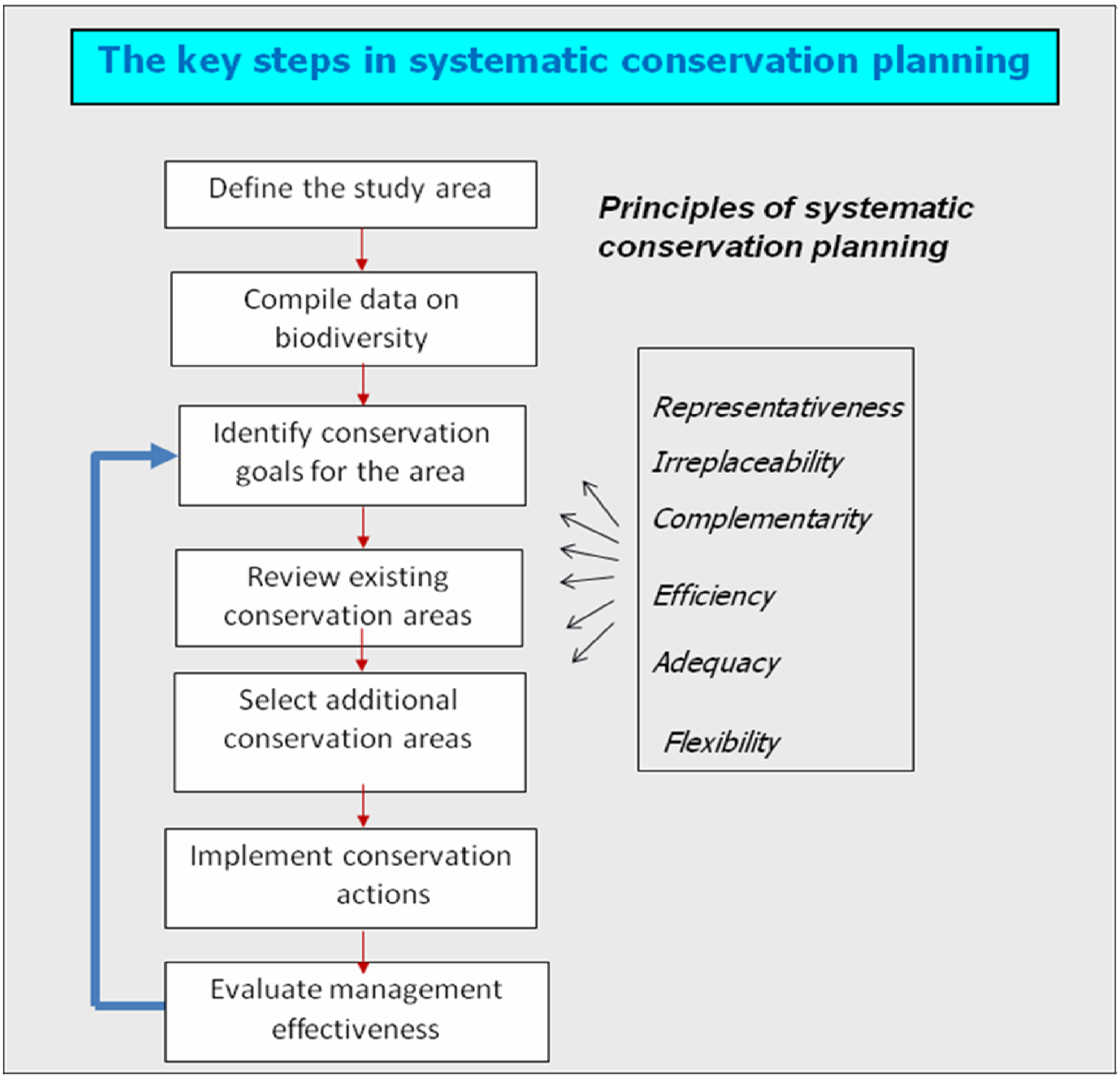

Figure 1.1. The various stages of systematic conservation planning and principles used to identify conservation priority areas. (Modified from Margules & Pressey 2000).

1.3 The application of systematic conservation planning approaches in freshwater ecosystems

Systematic conservation planning techniques that have been widely applied in terrestrial and marine ecosystems are yet to be fully adopted in freshwater ecosystems (Sarkar et al. 2006, Sowa et al. 2007, Linke et al. 2008). In addition, terrestrial biodiversity assessments seldom integrate freshwater biodiversity features when identifying priority areas for conservation (Nel et al. 2007, Linke et al. 2008). The exclusion of freshwater ecosystems from most terrestrial assessments is partly due to limited freshwater spatial data and and the complexity in planning for freshwater and terrestrial systems together (Abell 2002, Sowa et al. 2005, Suski & Cooke 2007). Freshwater ecosystems are connected laterally, vertically and longitudinally (Baron et al. 2002, Ward et al. 2002, Jansson et al. 2007), at a scale that is completely different from terrestrial ecosystems (Linke et al. 2008). In many cases freshwater and terrestrial biodiversity priority areas do not overlap making integration difficult. The development of techniques that seek to identify joint freshwater and terrestrial biodiversity priorities, while recognising their uniqueness has therefore been a major challenge (Abell 2002, Sowa et al. 2005, Nel et al. 2007).

The challenges to integration occur not only between freshwater and terrestrial systems but also between freshwater ecosystems themselves. The fact that freshwater ecosystems comprise rivers, wetlands, lakes and groundwater make their incorporation into a single conservation planning framework very challenging and yet integration is very important. This is important to ensure efficiency in the spatial design of biodiversity priority areas, and to minimise trade-offs between different components of the ecosystem. For example, wetlands which act as a transition between freshwater and terrestrial ecosystems, have not been fully integrated into freshwater conservation planning frameworks (Nel et al. 2008). Most conservation assessments treat wetlands as part of terrestrial ecosystems (e.g. Lombard et al. 1997, Madsen & Clausen 1998). The shortfall in this approach is that in most cases, the diversity of wetland types is not captured because the entire wetland system is treated as a single vegetation type (e.g. Lombard et al. 1997). And in many cases wetlands have been incorporated into conservation assessments only because of their importance as habitat for terrestrial species such as water fowl (Madsen & Clausen 1998, Linke et al. 2008). Failure to incorporate the diversity of wetland types will lead to inadequate wetland protection, because different wetland types support unique biodiversity, functions and benefits (Hauer & Smith 1998, Roe & Georges 2007, Standa & Ehrenfeld 2008).

The effective implementation of conservation plans is particularly challenging for freshwater ecosystems where the jurisdiction for the protection of freshwater biodiversity is multi-sectoral (Pahl-Wostl et al. 2007). In South Africa for example, the Water Act 1998, mandated the setting up of statutory bodies referred to as Catchment Management Agencies (CMA) and informal Water User Associations (WUA) to manage water resources, under the auspices of the Department of Water Affairs and Forestry (DWAF) (Mackay & Ashton 2004). Other departments such as that of minerals and agriculture also have a stake in the management of water resources. This has resulted in a complex institutional framework for the management of freshwater ecosystems (Mackay & Ashton 2004), and sectoral policy conflicts in South Africa (Roux et al. 2008). Implementing freshwater biodiversity priorities is therefore very challenging because no single institution is solely responsible for the protection of freshwater ecosystems. In order to effectively implement conservation priorities therefore requires significant level of coordination and cooperation between the different agencies both at the national and sub-national level (Roux et al. 2008b). Conservation planners also need to take into cognisance the importance of stakeholder mobilisation to facilitate the implementation of conservation plans (Knight et al. 2006). At present conservation plans are mostly designed by biologists with minimum input and/or interaction with social scientists and the public (Schwartz 2006). Conservation managers also often find conservation plans difficult to interpret or inaccessible and mostly rely on their experiences to make decisions (Pullin & Knight 2005)

In order to develop robust conservation strategies freshwater ecosystems need to be treated as social ecological systems, which are complex and dynamic, requiring interventions that detect changes in their pattern and are able to respond accordingly (Bohensky 2006). Because of the complexity of freshwater ecosystems, effective conservation strategies should take an interdisciplinary approach in developing successful solutions to solving freshwater problems (Dollar et al. 2007). Integrated water resource management (IWRM) approaches have attempted to incorporate the multiple facets of water resources (Jeffrey & Geary 2006), but most of the emphasis is on understanding freshwater ecosystem as a utilitarian resource. Less focus has been placed on incorporating biodiversity issues in developing management strategies (Gilman et al. 2004), as a result the condition of freshwater ecosystems continue to decline even though IWRM approaches have been widely applied globally (Jeffrey & Geary 2006).

1.4 Aims of this thesis

The issues discussed above highlight some of the challenges encountered in developing effective freshwater conservation strategies. In the last two years a tremendous amount of work has been done on freshwater conservation in attempts to address some of the challenges, such as accounting for longitudinal connectivity (Linke et al. 2007), ecological integrity (Mattson & Angermeier 2007), freshwater biogeographic framework (Abell et al. 2008), and planning for freshwater biodiversity persistence (Nel et al 2008).

This thesis aims to contribute to the body of work on freshwater conservation planning by developing tools and approaches for effectively incorporating different freshwater ecosystem types, aligning freshwater and terrestrial biodiversity priorities, and assessing the efficacy of implementing freshwater biodiversity priorities. I deliberately chose to ask a range of questions along the conservation planning continuum from biodiversity assessment to implementation, because of its usefulness in inferring broad lessons on freshwater conservation planning. This approach also helps to strengthen the links between the different conservation planning phases (Fig. 1.1), as it has already been acknowledged that conservation planners tend to focus on conservation assessments without dealing comprehensively with the challenges inherent in implementation (Knight et al. 2006).

1.5 Key questions

This thesis asks pertinent questions on freshwater conservation planning that are central to developing effective strategies for biodiversity conservation. These are listed in Table 1.1 and addressed issues from along the systematic conservation planning continuum from biodiversity assessment to implementation.

1.6 Structure of the thesis

The thesis is structured in such a way that each chapter is independent but the main themes being investigated are interlinked (Fig. 1.2). The first chapter is a general introduction, Chapter 2 is a focused review that identified the key gaps in systematic conservation planning, and is the basis for the subsequent questions addressed by the other chapters. The approach that was developed for wetland automation (Chapter 3) is used as a basis for classifying wetlands that were incorporated in the case study (Chapter 4). Chapter 5 attempts to align freshwater and terrestrial priorities to identify an efficient network of priority areas that comprehensively integrate both freshwater and terrestrial biodiversity. The resultant outcome of the priority areas is used to infer management effectiveness in the implementation of freshwater priority areas (Chapter 6).

Table 1.1. A breakdown of the chapters showing the theme in each chapter and key questions

Theme	Key questions
1) The status of freshwater conservation planning	a) Do conservation assessments integrate freshwater and terrestrial biodiversity in conservation planning? b) What measures are taken to ensure freshwater biodiversity persistence?
2) Automating wetland classification for conservation planning	c) How can wetland classification be effectively automated using basic readily available data for the purpose of conservation planning?
3) Planning for freshwater biodiversity persistence: A cases study from the Cape Floristic Region (CFR)	d) Where are the important freshwater biodiversity priority areas in the CFR? e) To what extent are the targets of different freshwater ecosystems achieved? f) What is the overlap between freshwater and terrestrial priority areas in the CFR?
4) Integrating freshwater and terrestrial priorities in conservation planning	g) Is the separate assessment of freshwater and terrestrial systems an efficient way of identifying biodiversity priorities for conservation? h) How can freshwater and terrestrial priorities be effectively integrated in conservation planning?
5) Assessing management effectiveness in implementing freshwater conservation priorities	i) What are the strengths and weaknesses in the current mechanisms for protecting freshwater ecosystems in South Africa?

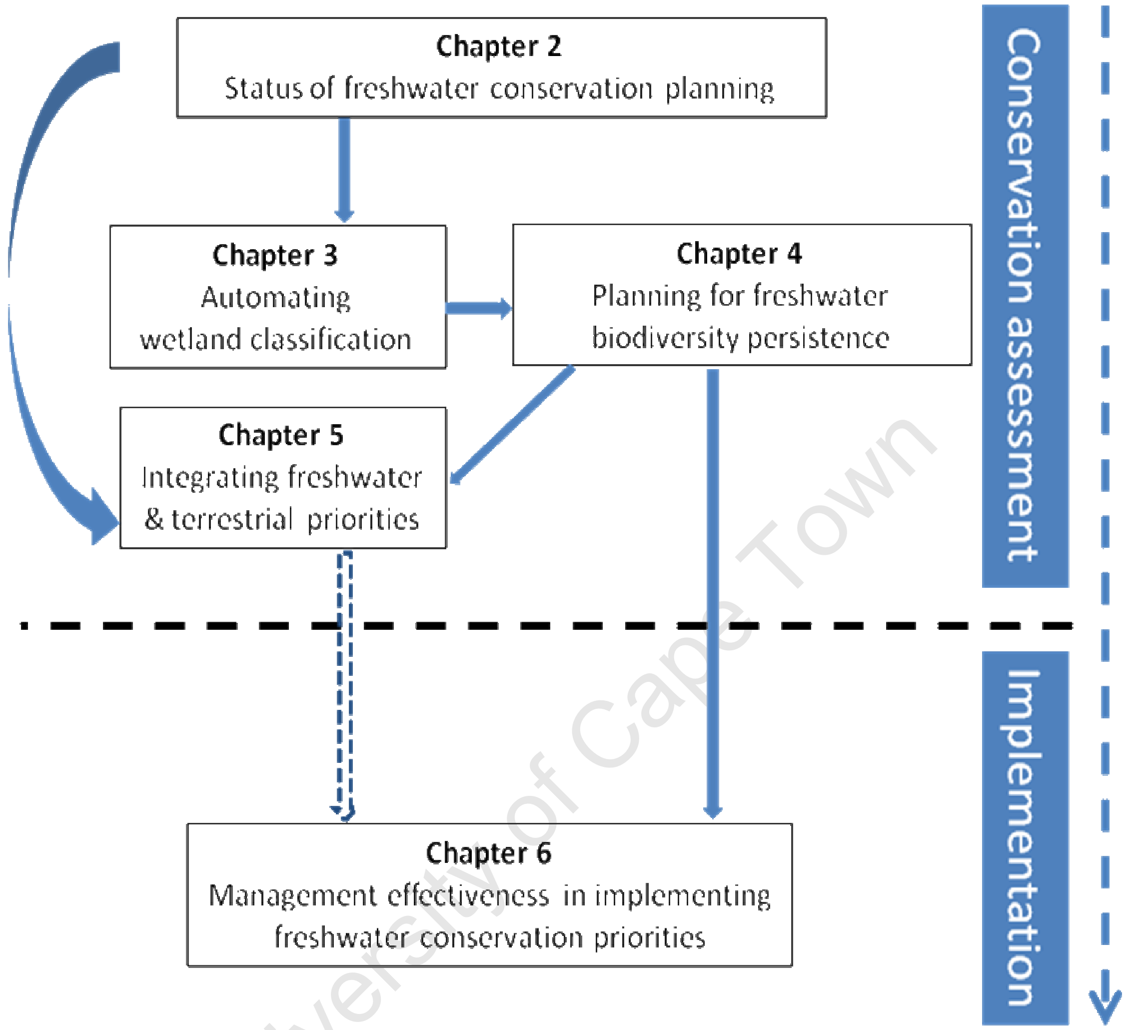

Figure 1.2. *A schematic representation of how each of the chapters in this thesis are related to each other and their position in the conservation planning continuum from biodiversity assessment to implementation*

1) What is the status of freshwater conservation planning? (Chapter 2)

This chapter sets the scene for the thesis, by undertaking a focused review of a few key journals to highlight some of the critical issues in freshwater conservation planning. It looked at how freshwater conservation planning has progressed over the last two decades in comparison with terrestrial conservation planning. The review focuses specifically on the extent to which freshwater and terrestrial biodiversity features have been integrated in conservation planning, the specific freshwater features assessed, and the extent to which issues of freshwater biodiversity persistence were addressed in conservation planning. The review identified some pertinent issues in freshwater conservation planning, such as the lack of integration of freshwater and terrestrial biodiversity priorities, incorporating wetlands into freshwater conservation planning and addressing implementation challenges. The issues raised in the review in turn informed the formulation of the research questions in the subsequent chapters of this thesis.

2) ***How can the diversity of wetland types be incorporated in freshwater conservation planning? Developing and testing an automation system for wetland classification. (Chapter 3)***

   Most of the efforts in freshwater conservation planning have focused on rivers, with very minimal consideration of wetland ecosystems. This is mainly because wetlands occur at the transition zone between freshwater and terrestrial ecosystems. It is still debatable whether it is more appropriate to incorporate wetlands with terrestrial ecosystems or rivers during conservation planning. However, one of the key challenges in assessing wetlands is their classification into different wetland types with relevance to systematic conservation planning. Because systematic conservation planning is often undertaken at a regional scale it involves the use of large datasets. As a result the classification of wetlands to capture their diversity has become a major impediment, because most classification frameworks are only appropriate for assessing individual wetlands but not a population of wetlands. In this chapter a geographical information systems (GIS) procedure is proposed for automating wetland classification. The proposed approach was used to classify wetlands mapped at a national scale, and the output of the classification was compared to wetlands that have been accurately mapped at fine scale.

3) ***How can we effectively plan for freshwater biodiversity persistence? A case study from the Cape Floristic Region, South Africa (Chapter 4).***

   This chapter is a demonstration of how planning for freshwater biodiversity persistence can be undertaken. It highlights some of the issues raised in the previous chapters such as the incorporation of wetlands into freshwater conservation planning frameworks, how to assess freshwater ecological integrity and connectivity of freshwater systems. A comparison is also done between freshwater and terrestrial biodiversity priorities to explore the trade-offs in areas of overlap. This study was undertaken in the Cape Floristic

Region (CFR), which has had a long history of terrestrial conservation planning. The freshwater conservation plan identified the gaps in the protection of freshwater ecosystems, and more specifically the extent to which terrestrial conservation plans in the implementation stage have addressed freshwater concerns.

4) ***How can freshwater and terrestrial biodiversity priorities be optimally integrated in conservation planning? (Chapter 5)***

The integration of freshwater and terrestrial biodiversity priorities remains a major challenge in conservation planning, due to the differences in their structure and function even though they are intricately connected. Due to the difficulty in integrating freshwater and terrestrial priorities in conservation planning, integrated approaches are rarely undertaken. This study represents one of the few attempts to use a decision support system (DSS) to integrate freshwater and terrestrial biodiversity priorities in conservation planning. The proposed protocol involves the separate assessment of freshwater priority areas, and using the outcome to influence the selection of terrestrial priority areas. This allows the conservation goals of both freshwater and terrestrial biodiversity to be adequately achieved without compromising their unique attributes.

5) ***Are there enabling mechanisms for the effective implementation of freshwater biodiversity priorities in South Africa? (Chapter 6).***

The previous chapters mostly address issues mostly concerned with biodiversity assessment with little emphasis on implementation side of the conservation planning continuum. This chapter therefore tries to build the link between biodiversity assessment and the implementation of biodiversity priorities, based on the premise that lack of attention to implementation challenges will impede the achievement of freshwater conservation goals. No conservation strategies can be successful without understanding the suitability of current mechanisms in the implementation of biodiversity priorities. In this chapter the opinions and perceptions of conservation practitioners mandated with the protection of freshwater ecosystems were sought in order to evaluate the extent to which products generated in systematic conservation planning are being used for decision making by implementing agencies. The evaluation took place in an interactive workshop environment with representatives of key implementing agencies in a key Water Management Area (WMA) in South Africa. The emphasis of the evaluation was on establishing social learning among keep partners that will lead to cooperation, and the achievement of freshwater conservation goals.

6) Synthesis (Chapter 7)

This Chapter discusses the key findings of this thesis, by synthesizing the issues dealt with in the previous chapters, and highlights some of the lessons learnt and the way forward.

CHAPTER 2. A REVIEW OF THE EXTENT TO WHICH FRESHWATER ECOSYSTEMS HAVE BEEN INCLUDED IN CONSERVATION PLANNING.

ABSTRACT

Systematic conservation planning has become an important tool in making sound conservation decisions. It is widely applied in the terrestrial and marine systems in the design of conservation priority areas. However, its application in freshwater systems is still relatively limited, and the effectiveness of current approaches in securing freshwater systems is not well studied. In this paper, we reviewed the effectiveness of conservation planning approaches in addressing freshwater issues based on a focused survey of 126 publications. Effectiveness was evaluated based on three main factors, (1) the extent to which freshwater and terrestrial priorities have been integrated in conservation planning, (2) the freshwater biodiversity features incorporated in conservation assessments, and (3) how issues of freshwater biodiversity persistence were addressed. We found that 75% of conservation plans were exclusively designed for terrestrial biodiversity without consideration of freshwater biodiversity whatsoever, 20% incorporated some aspects of freshwater and only 5% were designed for freshwater biodiversity. Most of the studies (58%) that considered freshwater biodiversity in their assessments did so because they were important for terrestrial species. There was huge disparity on the inclusion of measures to ensure freshwater biodiversity persistence. We conclude that, there is an urgent need to develop systematic conservation planning approaches and techniques for effective integration of freshwater and terrestrial priorities, and to effectively plan for freshwater biodiversity persistence.

2.1 Introduction

Freshwater systems are vital to humans and the environment; they provide us with ecosystem services such as food, drinking water, transport and flood regulation (Wiens 2002, Linden 2004, Dudgeon et al. 2006, Norris et al. 2007). But freshwater biodiversity are globally threatened due to human disturbance (Saunders et al. 2002). For example in N. America, it has been estimated that freshwater animals will face extinction at a rate five times greater than that of terrestrial animals and three times that of coastal marine animals (Ricciardi et al. 1999). Yet freshwater systems have been accorded the least protection compared to terrestrial and marine systems (Saunders et al. 2002, Kingsford & Nevill 2005). So why are freshwater systems not receiving the protection they deserve?

Traditionally freshwater systems have been considered as part of the terrestrial landscape. In this view freshwater systems are an element of the terrestrial landscape in the same way as vegetation, forests, roadways or urban centres. This view fails to consider that freshwater systems are functional landscapes in their own right, and are characterised by energy flows, boundary exchanges and organisms (Wiens 2002). Failure to treat freshwater systems as landscapes has negative implications for the conservation of freshwater biodiversity. It may have led to the incorrect assumption that protecting the terrestrial landscape will automatically secure the freshwater systems therein. As a result the conservation of freshwater systems often lags behind that of terrestrial systems (Moyle & Randall 1998). Indeed several studies have found that protected areas do not offer adequate protection to freshwater systems (Barret & Ansell 2003, Mancini 2005, Roux et al. 2008).

With regards to how different taxonomic groups and ecosystems are treated in conservation planning, Tear et al. (2005) stated that "this tendency toward generalisation has to be tempered by the realisation that what works for plants may not work for animals, and what works for populations may not work for systems". The same can be said of conservation assessments that assume that designating terrestrial priority areas will accord adequate protection to freshwater systems. Freshwater biodiversity are faced with unique challenges from their terrestrial counterparts, which requires them to be given special consideration. For example due to the open and unidirectional nature of river systems, aquatic organisms range widely and such movements may subject them to various stresses in their habitat (Dudgeon et al. 2006). The integrity of freshwater systems is also affected by land use activities at the catchment level compared to terrestrial biodiversity that are mostly affected by local land use activities (Amis et al. 2007).

The *ad hoc* delineation of reserve networks (Pressey 1994, Margules & Pressey 2000) paid little attention to freshwater systems and this has exacerbated the challenge of freshwater conservation. It has led to the call for conservation strategies that are specifically geared towards freshwater systems (Saunders et al. 2002, Kingsford &Nevill 2005, Abell et al. 2007, Suski & Cooke 2007), and the need for freshwater systems to be defined both in a systems context (Baron et al. 2002), and as landscapes in their own right (Wiens 2002).

In this chapter the effectiveness of conservation planning in addressing freshwater issues was reviewed by:-

- Assessing the extent to which freshwater and terrestrial priorities have been integrated in current conservation planning
- Evaluating what freshwater biodiversity features are included in conservation plans, and
- Assessing the consistency with which issues related to freshwater biodiversity persistence have been addressed in conservation planning.

2.2 Data sources

This review was based on an analysis of papers in three main conservation biology journals, which comprised *Conservation Biology*, *Biological Conservation* and *Biodiversity and Conservation*. These key publications are read by the mainstream conservation community (Abell 2002), and were among the five top journals in which conservation planning papers where published (1995- 2006), accounting for 40% of all conservation planning papers from a pool of 145 journals that published papers on conservation planning. The period between 1995- 2006, is a critical one in the advance of systematic conservation planning that included the first publication that succinctly defined the principles of systematic conservation planning by Margules & Pressey (2000). It was assumed therefore, that a focused review of these journals would yield a good sample size to examine current trends in systematic conservation planning.

Keyword search terms were used to retrieve articles from the ISI Web of Science (http://www.newisiwebofknowledge.com) in March 2007. The keyword search terms used to retrieve the articles were "conservation assessments", "conservation planning", "conservation plan", "conservation evaluation", "conservation value", "reserve selection", "area selection", "area identification", "priority area", "bioregional conservation", "bioregional planning", "ecoregional assessment", "ecoregional conservation", "integrated conservation" and "natural areas identification". In addition to the above approach, I perused manually through all issues of *Conservation Biology* and *Biological Conservation*, since their first publication in1987 and 1968 respectively to the last issue published in 2006, to validate the search terms used. This review was also updated with all the key publications on freshwater conservation planning that were published in the last two years since this review was first conducted.

2.3 The review process

Articles were scanned by reading the title and abstract with further perusal of the entire paper if the article was deemed relevant. For an article to qualify for review, it must have explicitly stated objectives related to biodiversity conservation. Studies that did not lead to a prioritisation of areas for conservation (e.g. Knight et al. 2006a), those that used hypothetical datasets (e.g. Moilanen & Cabeza 2005), and review articles (e.g. Leslie 2003) were not included in our analysis. Generally for a paper to qualify for this review, it had to have a spatial component to answer the 'where' question, and therefore included maps or at least the explicit identification of specific areas for conservation. Marine studies were excluded, except for a few that addressed terrestrial and/or freshwater issues in their assessments. Based on these criteria I identified 126 studies (database available on request) which formed the basis for this review.

The selected studies were classified according to the focus area of the assessment to identify the proportion of studies that gave due consideration to freshwater ecosystems. To answer the question of how effective freshwater issues have been addressed in conservation planning, I extracted data on the freshwater biodiversity features that were assessed e.g. the type of taxa or species. Further data was extracted on measures of freshwater biodiversity persistence such as connectivity, ecological integrity, catchment level considerations and ecological processes. Where possible, data was also collated on the geographic scale of the studies, the location, and the type of planning unit. It was acknowledged that, even though some studies may have been designed purely for academic purposes, systematic conservation planning is primarily about implementing sound conservation decisions (Margules & Pressey 2000, Knight et al. 2006b, Sarkar et al. 2006, Oetting et al. 2006); as a result it is important that conservation planners also address issues of implementation. For this reason we also documented the extent to which implementation issues were addressed in systematic conservation planning.

2.4 Trends in conservation assessments indicating the extent to which freshwater systems were included.

Over the last two decades, conservation planning has developed into a powerful tool for making sound conservation decisions (Fig. 2.1). Most of the effort however, has been focused on terrestrial systems. Whereas studies on terrestrial conservation planning have increased almost exponentially in the last two decades (Fig. 2.1), it was only in the last decade that we begun to see papers published on freshwater conservation planning. The numbers of studies that focus on freshwater conservation planning have remained relatively constant over the period, peaking only during the last year (2006) of the review.

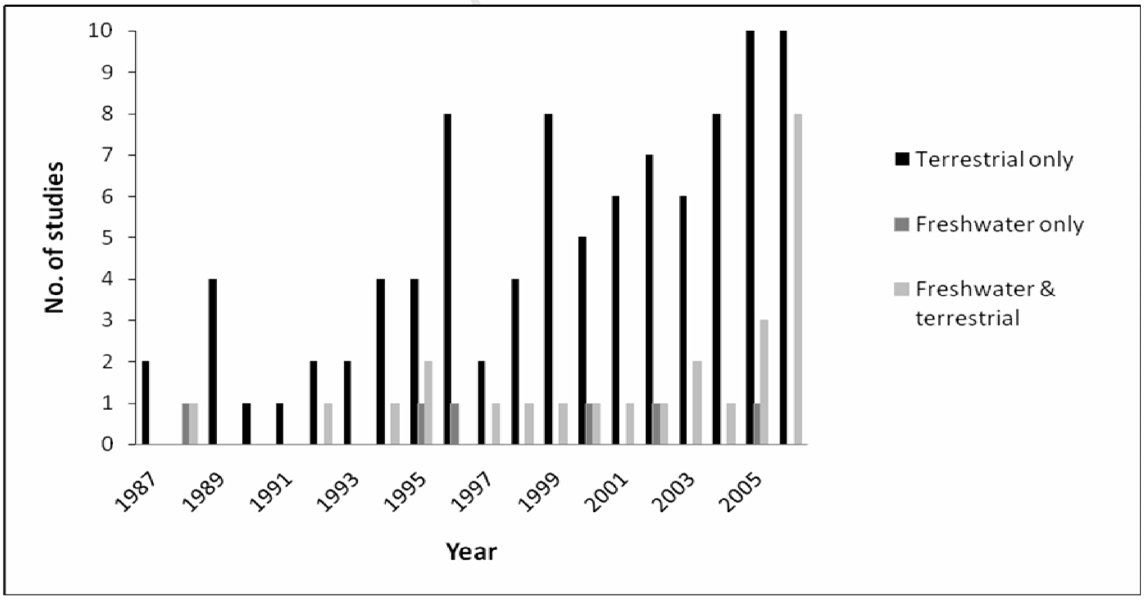

Figure 2.1. *Trends in systematic conservation planning over time. The graph shows the number of studies that identify conservation priorities for terrestrial biodiversity only, those that incorporate some aspects of freshwater biodiversity, and those studies designed specifically to identify priorities for freshwater biodiversity.*

The majority of conservation assessments were carried out at the regional scale, which accounted for about 73 (53%) of the studies reviewed (Fig. 2.2). Forty (29%) conservation assessments were carried out at the national scale, 12 (9%) at the local, and 10 (7%) at the continental scales. Only one of the studies was conducted at a global scale. The dearth of global conservation assessments could be attributed to the difficulty of carrying out sides at the global scale, where the derivation of uniform datasets is a major challenge. However, global assessments have previously been used successfully to define biodiversity priorities such as the global biodiversity hotspots (Myer et al. 2000).

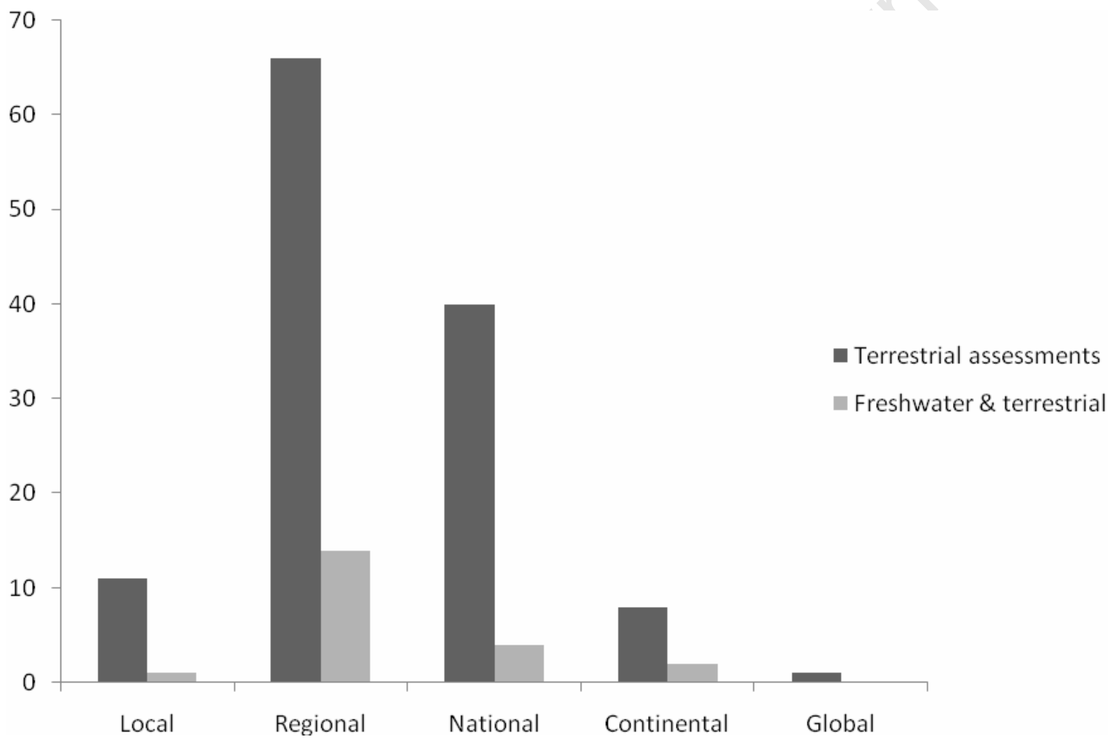

Figure 2.2. The scale at which conservation assessments are carried out for terrestrial biodiversity only, and when freshwater and terrestrial biodiversity are integrated in conservation assessments.

2.5 Do conservation assessments incorporate and freshwater biodiversity?

There was minimal incorporation of freshwater biodiversity in conservation planning, with 75% of conservation plans designed exclusively for terrestrial biodiversity. About 20% of conservation assessments integrated some aspects of freshwater biodiversity, and only 5% were exclusively designed for freshwater biodiversity. The bias of conservation planning towards terrestrial biodiversity is of major concern to biodiversity conservation, given the fact that freshwater biodiversity are more imperilled than terrestrial biodiversity (Ricciardi et al. 1999), and that terrestrial conservation strategies are not always effective in securing freshwater ecosystems (Mancini 2005, Roux et al. 2008). Integrating freshwater and terrestrial assessments in conservation planning is a potentially useful approach for achieving the conservation goals of both ecosystems (Chapter 5), and for ease of implementing conservation decisions (Nel et al. 2008). Integrated approaches to conservation planning do not appear to be widely used, despite been recognised as important (Abell 2002, Sowa et al. 2005, Nel et al. 2008). There are few examples where integration has been applied in conservation planning, such as preferentially selecting freshwater priority areas adjacent to protected areas (Thieme et al. 2007), or generating separate maps that show integrated priorities, and others that are ecosystem specific (Sowa et al. 2005). There are however, no examples that illustrate the pros and cons to integrated conservation planning, and the trade-offs incurred in this approach, this therefore represents a major research gap (Chapter 5).

2.6 To what extent were freshwater biodiversity represented in conservation assessments

I further reviewed the 21 studies that included some freshwater features or those that were designed exclusively for freshwater biodiversity. The aim of critically reviewing the freshwater related studies was to examine their effectiveness in addressing pertinent freshwater concerns that are deemed vital for their protection. I found no consistency in the extent to which freshwater biodiversity features or measures of persistence were included in conservation assessments (Fig. 2.3). Conservation plans that had a clear freshwater mandate did appear however, to be more comprehensive in their consideration of freshwater issues. There has also been considerable improvement over the years, on the extent to which important freshwater biodiversity features are incorporated in conservation planning.

Freshwater biodiversity features were generally incorporated in conservation planning in three main ways: -

- Freshwater systems as habitat for terrestrial species or a vegetation type
- Direct incorporation of freshwater taxa/species
- Physical classification of the aquatic habitat/system as surrogate for freshwater biodiversity

Most freshwater biodiversity features were included in conservation assessments because they were a habitat for specific terrestrial taxa such as water fowl (Madsen & Clausen 1998) or, in the case of wetlands, they were considered as a terrestrial vegetation type (e.g. Lombard et al. 1997). Of the 21 freshwater related studies, 52% included freshwater biodiversity in their assessment because of their importance to terrestrial biodiversity (Fig. 2.3). This finding may imply that the importance of freshwater biodiversity is still not fully recognised in conservation planning, or it is still assumed that by prioritising for the terrestrial landscape, freshwater biodiversity features will be secured. This approach undermines the basic principle of representativeness in systematic conservation planning (Margules & Pressey 2000), because it fails to recognise the inherent biophysical diversity in freshwater systems. For example if wetlands are assessed for their importance as habitat for terrestrial species (e.g. Madsen & Clausen 1998), little emphasis is placed on capturing the diversity of wetland types.

A reasonable percentage (48%) of the 21 freshwater related studies incorporated freshwater taxa directly in their conservation assessment, but most were restricted to a specific taxon. Fish and amphibians were the most common taxa included in conservation assessments (e.g. Sarakinos et al. 2001, Filipe et al. 2004). Other taxa also included in conservation assessments were fairy shrimp (Pyke & Fischer 2005), California salamander (Pyke 2006), and American crocodiles (Thorbjarnarson 2006). The poor incorporation of species information in conservation planning maybe attributed to lack of data on freshwater biodiversity, which are often minimal and extremely patchy (Thieme et al. 2007). Mapping freshwater biodiversity spatially is also very complex, in many conservation assessments therefore, the only option is to use readily available datasets.

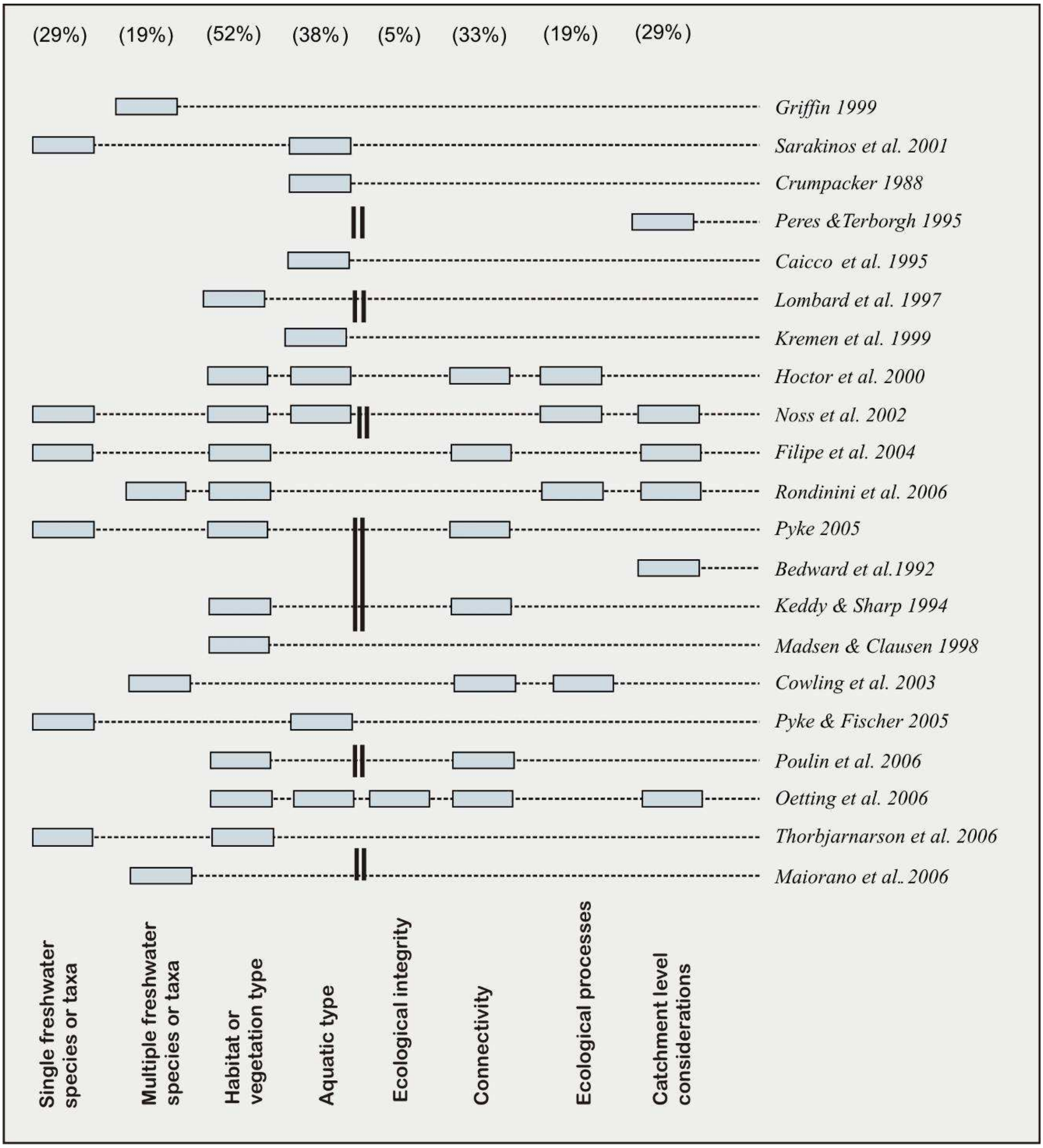

Figure 2.3. *Studies of systematic conservation planning (1987- 2006) that incorporated freshwater in their assessment, showing the freshwater attributes, measures of persistence and the percentage number of studies that included each criterion. The broken double vertical lines separate freshwater biodiversity features (left) and measures of persistence (right).*

About 38% of the 21 freshwater related studies incorporated freshwater biodiversity through the physical classification of the freshwater systems as a surrogate for biodiversity (Nel et al. 2007, Thieme et al. 2007) (Fig. 2.3). I also lumped into this category some studies that dealt with a specific freshwater system type such as wetlands or peatlands, but did not further classify them according to their different types. The number of studies that tried to capture the inherent diversity in freshwater systems was therefore probably much less than is reported here. Examples of physical classification of aquatic system/habitat included setting targets for headwater rivers (Kremen et al. 1999), seepage slopes and coastal wetlands (Oetting et al. 2006). Other classifications of rivers include wild and scenic rivers, and shellfish harvesting rivers (Hoctor et al. 2006). It should be noted that the physical classification of freshwater systems as a surrogate for biodiversity is not always effective. It is therefore important that surrogate data are used in conjunction with real biological data whenever possible.

2.7 What measures were taken to ensure persistence of freshwater biodiversity?

One of the main goals of systematic conservation planning is to ensure biodiversity persistence (Margules & Pressey 2000, Cowling et al. 1999, Nicholson et al. 2006). Designating a representative suite of conservation areas is therefore not enough, unless measures to ensure species persistence and their ecological processes are taken into consideration (Groves et al. 2002, Cowling et al. 2003, Lourie &Vincent 2004, Rouget et al. 2006). Incorporating measures of persistence in identifying priority areas however, remains a major challenge in conservation planning. The attempts to incorporate biodiversity persistence in conservation planning often results in trade-offs between achieving representativeness or incorporating measures of biodiversity for persistence, with persistence the ultimate loser (Rouget et al. 2006). The challenges of incorporating measures of freshwater biodiversity persistence in conservation planning were quite apparent in this review. The few studies that considered issues of freshwater biodiversity persistence did so in four main ways: -

- through connectivity of landscape patches or priority areas
- catchment-level considerations
- incorporation of ecological integrity
- mapping of ecological processes

Connectivity was considered in this respect as both landscape scale (terrestrial) connectivity and connectivity within freshwater systems. It was found that 33% of the 21 freshwater related studies attempted to build connectivity into their conservation assessment process. For example Cowling et al. (2003) used inter-basin riverine systems as corridors to link between priority areas, in an assessment that integrated both freshwater and terrestrial biodiversity. Connectivity is critical in restoring systems (Jansson et al. 2007), because it is crucial for sustaining their functional processes (Ward et al. 2002). In freshwater systems, it comprises of longitudinal, vertical and lateral connectivity (Baron et al. 2002, Ward et al. 2002, Jansson et al. 2007). Freshwater and terrestrial landscapes also need to stay connected because many species depend on both systems to complete their lifecycles (Hanet & Janovy 2002, Semlitsch 2002). Maintaining connectivity in the landscape is thus critical for species persistence and ecosystem functioning (Leslie 2005, Boulton 2007, Lake 2007).

Catchments are the primary landscape units in which speciation occurs (Moyle & Randall 1998). Land based activities at the catchment scale also influence instream ecological integrity (Amis et al. 2007), as a result catchments are important for devising effective conservation strategies. Catchments also play an important role in ensuring lateral connectivity within freshwater ecosystems, and their usage as planning units captures other catchment level processes critical for biodiversity persistence. In this study however, only 5% of the reviewed assessments used catchment delineations as a planning or management unit (e.g. Filipe 2004, Oetting et al. 2006, Rondinini et al. 2006).

The maintenance of high ecological integrity in freshwater systems is a key goal of conservation planning (Mattson & Angermeier 2007, Groves et al. 2002). Ecological integrity is important because conservation targets must be achieved in intact river systems to ensure persistence of biodiversity (Nel et al. 2008). Current conservation planning approaches rarely incorporate ecological integrity in their assessments however (Mattson & Angermeier 2007). I found only one study (Oetting et al. 2006) (Fig. 2.3) that incorporated ecological integrity by ensuring that only “high-quality” catchments were selected in the process of identifying priority areas. Perhaps this omission can be explained by the complexity associated with assessing the ecological integrity of freshwater systems in conservation planning (Amis et al. 2007, Nel et al. 2007). Most tools available for assessing ecological integrity are site-based, with very few available at the larger scale in which conservation planning exercises are performed (Linke et al. 2007). The exclusion of ecological integrity in conservation planning might also be related to the objectives of the assessments. Conservation plans that were designed for terrestrial biodiversity, for example, usually did not include measures of freshwater ecological integrity.

The inclusion of ecological processes in conservation planning is very difficult due to lack of appropriate spatially explicit surrogates, more so in freshwater systems. In this study we found that 19% of the 21 freshwater related studies attempted to include ecological processes in their assessment (Fig. 2.3), though most of the ecological processes assessed were not related to freshwater systems. Some studies (e.g. Noss 2002) stated the conservation of ecological processes as an objective, but did not state explicitly whether ecological processes were included in their assessment. Such studies were recognised as having included freshwater processes even though they were not explicitly mapped. Some studies such as Hoctor (1999) evaluated large natural areas as surrogates for ecological processes, while Filipe (2004) and (Rondinini et al. 2006) justified the capturing of ecological processes through the inclusion of connectivity and the use of catchments respectively. In general both instream processes and those associated with riparian areas were very poorly addressed in conservation planning.

2.8 Conclusions

Systematic conservation planning is still largely focused on terrestrial biodiversity. It is only recently that significant progress is starting to be achieved in freshwater conservation planning (Table 2.1). In the last two years since this review was conducted, significant progress has been achieved in advancing freshwater conservation planning. Some notable publications that have appeared in the recent past include: developing measures for incorporating connectivity in freshwater ecosystems (Linke et al. 2007), hierarchical planning strategies (Abell et al. 2007), assessing ecological integrity (Mattson & Angermeier 2007), and numerous conceptual analysis on how to effectively conserve freshwater biodiversity (e.g. Sloane et al. 2007, Nel et al. 2008, Roux et al. 2008).

There are however, some critical issues that have arguably not been well addressed in freshwater conservation planning, and require more attend, such as: (1) Developing more effective approaches for integrating wetland ecosystems into freshwater conservation planning frameworks (2) Incorporating, aquatic, riparian and catchment-scale ecological processes in conservation planning. (3) Exploring the concept of land-sea planning more comprehensively by developing effective techniques for aligning terrestrial, freshwater and marine biodiversity priorities. (4) Designing conservation plans with a strong implementation focus, by developing products that are user friendly and focus more on the process of conservation planning (stakeholder engagement), than outputs (maps).

There is a need to do more systematic conservation planning in some regions of the world such as sub-Saharan Africa (excluding South Africa) and Southeast Asia. Most conservation assessments for these regions were conducted at the continental scale with barely any regional or local scale assessments (Fig. 2.4). Most systematic conservation planning studies are still concentrated in North America, Australia and South Africa (Fig. 2.4), countries that have been credited with pioneering the practice of systematic conservation planning. The disparity in where conservation plans have been carried out is perhaps a reflection of the fact that some regions are poorly studied and resourced with a severe dearth of data for conservation planning. It should however, be noted that some of these regions are of global significance to biodiversity conservation, with numerous biodiversity hotspots. The large swathes of natural forests in these regions, albeit under severe threats are vital for mitigating global climate change, their significance can therefore not be underestimated.

There is still a glaring disparity between conservation assessments and implementation. Very few studies crossed that boundary between carrying out a conservation assessment and implementing those decisions. Although it was not easily discernible from this review whether the studies were carried out just as an academic exercise or they were meant to inform conservation action, it is disturbing that implementation issues were not adequately addressed or reflected in the literature. The effectiveness of conservation plans depends on the consideration of implementation issues from the onset of the assessment (Knight et al. 2006b). This is even more pertinent in freshwater systems, where water is a commodity with very high demand and multiple stakeholders. Therefore conservation planners cannot discount the importance of addressing implementation issues if such plans are to make a meaningful contribution to conservation.

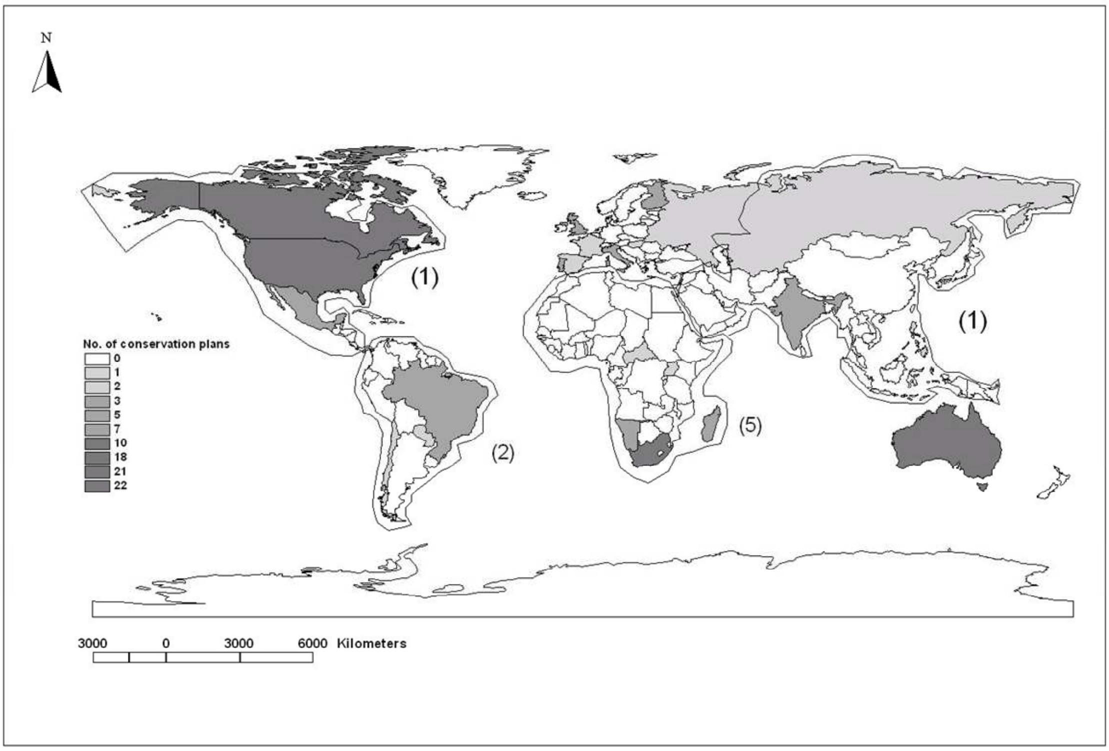

Figure 2.4. Global distribution of where most conservation assessments are carried out between 1987- 2006. The numbers next to some of the continents show the number of conservation assessments at the continental scale.

Table 2.1. Key freshwater conservation planning papers that were published in the last two years (2007 & 2008), showing their major area of focus

Main theme	**Primary objective**	**Key biodiversity features**	**Scale**	**Planning region size**	**Location**	**Reference**
Biodiversity assessment	Framework for river assessment	River ecosystems	National	1.2m km^2	South Africa	Nel et al. 2007
Biodiversity assessment	Indices for assessing ecological integrity	Various freshwater biota	Regional	3 m km^2	Australia	Norris et al. 2007
Conceptual analysis	Review of protected areas (PA)	Freshwater habitats & species	n/a	n/a	n/a	Suski & Cooke 2007
Conceptual analysis	PA for freshwater ecosystems	Freshwater habitats & species	Various scales	n/a	n/a	Abell et al. 2007
Conceptual analysis	Wetland inventory, assessment and monitoring	All wetlands	n/a	n/a	n/a	Davidson & Finlayson 2007
Conceptual analysis	Call for more effort to conserve freshwater biodiversity	Freshwater habitats & species	Various scales	n/a	n/a	Vance-Borland et al. 2008
Conceptual analysis	Implementing freshwater conservation strategies	Freshwater habitats & species	n/a	n/a	South Africa	Roux et al. 2008
Conceptual analysis	Linking land to sea conservation	Terrestrial, freshwater and marine biodiversity	n/a	n/a	n/a	Sloane et al. 2007
Conceptual analysis	Assessing the progress and challenges in freshwater conservation planning	n/a	n/a	n/a	n/a	Nel et al. 2008
Environmental classification	Developing measures for environmental classification	Fish communities	regional	Not specified	New Zealand	Snelder et al. 2007

Main theme	Primary objective	Key biodiversity features	Scale	Planning region size	Location	Reference
Freshwater classification	Freshwater classification	Freshwater fish	Global	149m km^2	n/a	Abell et al. 2008
GAP analysis	PA for freshwater ecosystems	Water Beetles	Local	18815 km^2	Iberian Peninsula	Abellan et al. 2007
GAP analysis	PA coverage of freshwater biodiversity	Freshwater habitats & species	Regional	57,00km^2	South Africa	Roux et al. 2008
Gap analysis	Effectiveness of PA in conserving freshwater biota	European eel, wetland	Local	7000 ha	France	Cucherousset et al. 2007
Priority areas	Mapping and prioritizing wetlands	Palustrine and estuarine wetland types	Regional	Not specified	New Zealand	Ausseil et al. 2007
Priority areas	Assessing ecological integrity	invertebrates	Regional	227,600km^2	Victoria, Australia	Linke et al. 2007
Priority areas	Gap analysis and identification of priority areas	All riverine ecosystems	Regional	Not specified	USA	Sowa et al. 2007
Priority areas	Method for river conservation value	Benthic macro-invertebrates	Regional	227,600km^2	Victoria, Australia	Linke et al. 2008
Priority areas	Assessing ecological integrity	All riverine ecosystems	Regional	55,400km^2	Tennessee, USA	Mattson & Angermeier 2007
Priority areas	New method for identifying priority areas	Fish species	Local	2.5m ha	North Island, New Zealand	Moilanen et al. 2007
Priority areas	strategies for freshwater conservation in data-poor regions	Abiotic surrogates for freshwater biodiversity	Regional	160,00km^2	Peru & Bolivia	Thieme et al. 2007

CHAPTER 3. AN AUTOMATED APPROACH FOR CLASSIFYING WETLANDS USING GIS FOR CONSERVATION PLANNING

ABSTRACT

Wetland classification is important for conservation planning to incorporate the diversity of wetland functions when identifying priority areas for biodiversity conservation. However, most wetland classification frameworks are only suitable for classifying individual wetlands, making their application in conservation planning challenging. In South Africa a wetland classification system has been developed, but it's effectiveness in classifying a large group of wetlands has not been tested. In this study an approach was developed for automating wetland classification for the purpose of conservation planning. The approach was used to classify wetlands mapped at a national scale, and the output of the classification was compared to wetlands that have been mapped at fine scale. The results showed that the proposed framework is robust for classifying wetlands. Level 3 in the hierarchical approach produced the most accurate classification of wetlands, at a scale most suitable for conservation planning. Wetlands at level 3 were classified with a high level of accuracy ranging from 50% to 90% for the 4 wetland types identified at broad scale. At a higher level of the classification hierarchy however, the results were inconsistent implying that the scale of analysis has an influence on the accuracy of wetland classification. The extent to which wetlands can be classified should therefore depend on the objectives of the classification and the accuracy of the available data.

3.1 Introduction

Wetlands are important freshwater ecosystems, where they provide essential ecosystem services such as flood and water quality regulation, groundwater discharge/recharge, and carbon sequestration (Ausseil et al. 2007, Cassidy 2007). Wetlands are also important habitat for species, and have been shown to provide a more suitable habitat for numerous taxa than upland areas (Richter & Azous 2001). However, they are among the most threatened aquatic ecosystems (Ortega et al. 2004). Wetland degradation results directly in biodiversity loss and the associated ecosystem services that wetlands provide. But wetland conservation is very challenging compared to the conservation of other terrestrial ecosystems, where for a long time they have been regarded as wastelands (Kim et al. 2006). This may be attributable to various factors among which include: the landscape location of wetlands that predisposes them to waste disposal (Dudgeon et al. 2006), difficulty in assessing their ecological integrity (Ortega et al. 2004), and the lack of appropriate classification approaches for incorporating wetlands into conservation planning frameworks (Chapter 2).

Systematic conservation planning has made major strides in the last decades in enhancing the conservation of terrestrial biodiversity (Chapter 2), and recently major progress has been attained in the conservation of freshwater ecosystems (Nel et al. 2008). Most of the focus is still on rivers however, and the challenges in wetland conservation have largely not been well addressed. For conservation planning techniques to be effectively applied in wetland conservation, there is a need for the development of tools to assess wetlands. There is an urgent need for the development of techniques for the rapid assessment of wetland integrity and wetland classification frameworks that are suitable for conservation planning. Wetland classification is important because different wetland types perform different functions (Hauer & Smith 1998, Standa & Ehrenfeld 2008). Since the essence of systematic conservation planning is to plan for a comprehensive, adequate and representative set of biodiversity priority areas (Margules & Pressey 2000), it is imperative that the diversity of wetlands is taken into consideration during planning so that each wetland type is adequately represented and protected.

Wetland classification is the process of grouping wetlands according to the similarity in their physical characteristics, chemical or biotic composition (Pressey & Adam 1995, Zoltai & **Vitt**

1995). The classification is primarily based on the processes that control wetland function (Zoltai & Vitt 1995). Wetland hydrology, water source and hydrodynamics are the most important determinants of wetland function (Mitsch & Gosselink 2000). As a result hydrogeomorphic (HGM) classification is the most common approach to classifying wetlands. The HGM is based on hydrological segregation of wetlands according to their function and benefits (Nielson et al. 2006). Based on the two main parameters of landscape position (hilltop, slope, valley bottom, and coastline) and dominant water source, wetlands can be classified into various types. However, due to the complexity of wetlands and differing objectives for classification, there are no universally applicable wetland classification systems (Smith et al. 1995, Zoltai & Vitt 1995).

In South Africa a wetland classification approach was first proposed 30 years ago (Noble & Hemens 1978), where wetlands were categorised broadly into six classes of inland waters, artificial impoundments, pans and lakes, coastal and estuarine lakes, estuaries, and estuarine lagoons. A major exercise was undertaken a few years ago to establish a standard wetland classification system for South Africa (Ewart-Smith et al. 2006), and this is currently being refined to make it more explicit. The most basic elements of the wetland classification framework use a hydrogeomorphic approach to classify wetlands, which comprise freshwater ecoregions, hydrology, landscape position, landform and fluvial integration (Ewart-Smith et al. 2006). The classification system is hierarchical in structure ranging from the most general, subsystem, to functional structure and habitat units. Each level of the hierarchy is defined by discriminators, the main ones being connectivity to open ocean, drainage, landform and tidal regime, dominant cover types and dominant life-form characteristics (Ewart-Smith et al. 2006). Systems are the most general level of classification, where marine, estuarine and inland ecosystems are segregated based on their connectivity to the open ocean. The next level of classification is that of subsystems, which incorporates the regional setting of the wetland by distinguishing wetlands based on ecoregion setting and their vegetation grouping. The third level of classification is based on functional units, and landscape position is the primary discriminator. The final level in the hierarchy is based on hydrogeomorphology, where landform and hydrology are the main discriminators.

The hierarchical approach adopted in the wetland classification framework for South Africa is very useful, because it allows wetlands to be classified into differing details. The major shortfall of the classification system for conservation planning is that some of the discriminators used are impractical to implement such as the 'fluvial integration'. The incorporation of the regional setting should also have been in the last tier in the framework, to simplify the process. A similar problem has been noted with the Ramsar Classification system, where Semenuik & Semenuik (1997) propose that the criterion on geomorphic setting should precede biological attributes. As it stands now the proposed classification system for South Africa is more suited for classifying individual wetlands, but may prove to be daunting when used to classify a large number of wetlands.

There is a need to therefore develop robust approaches for implementing the proposed wetland classification system. This will enable the identification of priority wetland systems in South Africa and to assess their biodiversity, functions and benefits. This study hopes to achieve this by developing an approach for automating wetland classification for conservation planning using readily available map scale data and geographical information systems (GIS). The main issues being addressed are:-

- The development of a GIS protocol for automating the current wetland classification system developed for South Africa
- Applying the classification framework to a region in South Africa
- Comparing the broad wetland classification with wetlands mapped at a fine-scale for the region.

3.2 Method

3.2.1 The wetland automation and classification framework

The first step in testing the wetland classification system required the development of rules and criteria for automating the classification (Fig. 3.1). Automated wetland classification is vital for conservation planning because of the broad scale of analysis and the large datasets involved. To incorporate the diversity inherent within wetland systems therefore requires the use of a simple but robust approach for classifying wetlands. Consequently one of the important criteria for testing the suitability of a national wetland classification system is whether it can be fully be automated using tools such as geographical information systems (GIS). In this study several factors were thus taken into account in implementing the wetland classification system with relevance to conservation planning:-

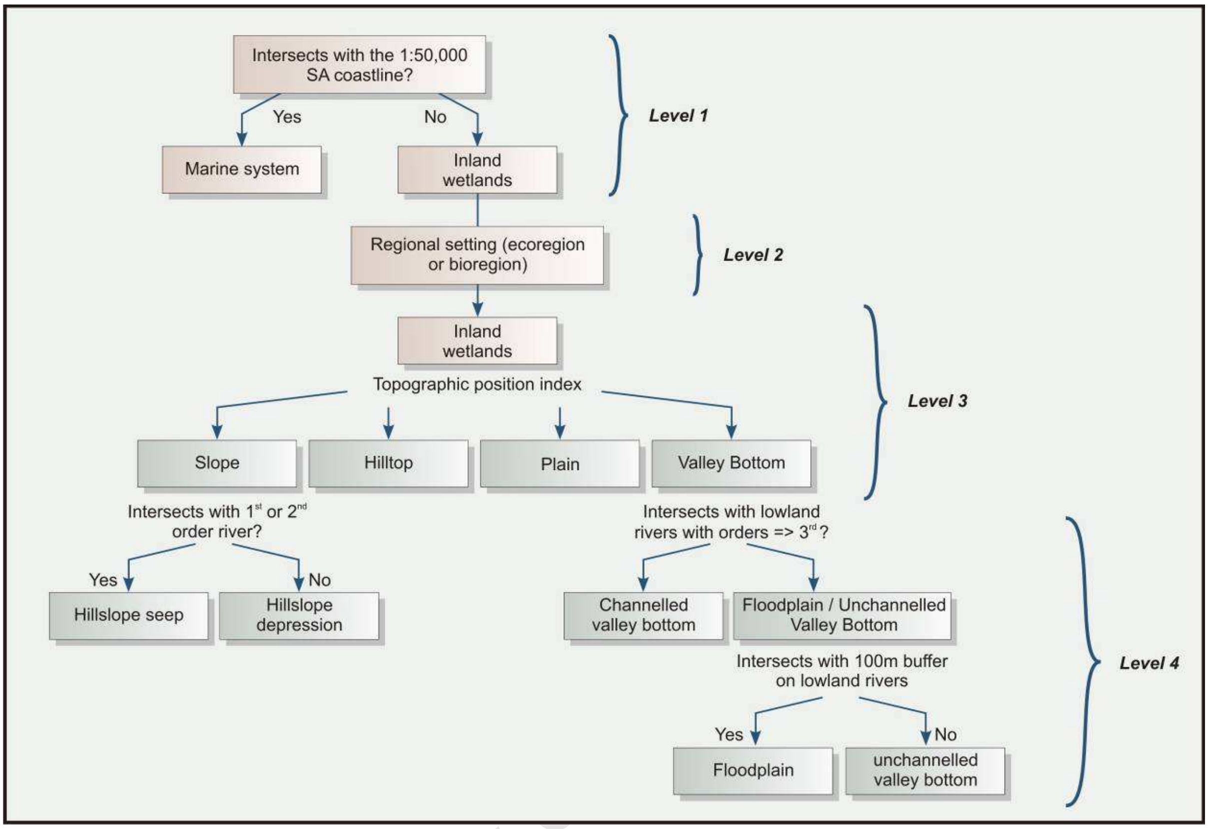

Figure 3.1. Criteria for automating wetland classification, showing the different levels of the classification that can be implemented.

- The approach should be simple and implementable on a GIS platform
- The Criteria must be based on readily available data at a broad scale
- The approach should enable comparison between wetlands that have been mapped at various scales on a pixel by pixel basis to minimise delineation errors.
- It should comprise a hierarchical protocol with various levels of classification, to incorporate both the geomorphic and hydrologic determinants of wetland types
- Each level of the automation framework should be independent and give rise to a wetland type, which can be further split into sub-classes

3.2.2 Implementing the wetland classification framework using a rule-based GIS approach

Based on the above criteria, GIS based rules were developed to automate the wetland classification developed for South Africa (Table 3.1). The approach uses very basic information, while acknowledging that these may have limitations on the extent to which wetlands can be classified. The aim was to enable a coarse wetland classification that will result in at least four main wetland types as specified in the wetland classification system (Table 3.1).

The four major wetland types in level 3 of the classification framework can be derived based solely on landscape position. Further sub-divisions using hydrology could be used to obtain a full hydrogeomorphic classification resulting in three additional wetland classes (floodplains, channelled and unchannelled valley bottom). However, due to the large scale of analysis and the coarse datasets, it is hypothesised that the finer the classification the less the confidence in the accuracy of classifying the wetlands. For broad-scale conservation planning purposes therefore, it is proposed that wetland classification be based only on geomorphic characteristics unless the hydrological data is sufficient for wetland classification to be achieved with a high level of confidence.

The automation framework comprised four levels which are in a hierarchical order. Each level of the hierarchy corresponds to the wetland classification system developed for South Africa.

- Level 1: discriminates between marine system and estuarine/inland wetland systems
- Level 2: Incorporates the regional setting as specified in the wetland classification system using ecoregions or bioregions based on vegetation types (Table 3.1)
- Level 3: discriminates between different inland wetland systems based on topography
- Level 4: discriminates between riverine and non-riverine wetlands

*Table 3.1. The South African wetland classification system showing the different wetland classification levels (**Ewart-Smith et al. 2006,** modified by **SANBI 2009**).*

LEVEL 1: SYSTEM	*LEVEL 2: REGIONAL SETTING*		*LEVEL 3: LANDSCAPE UNIT*	*LEVEL 4: HYDROGEOMORPHIC UNIT*		
CONNECTIVITY TO OPEN OCEAN	*ECOREGION*	*BIOREGION*	*LANDSCAPE SETTING*	*LANDFORM, LANDSCAPE POSITION & HYDROLOGY*		*DRAINAGE*
	A	B		A	B	C
INLAND	DWAF Level I Ecoregions	Bioregions of National Vegetation Map (Mucina & Rutherford 2006)	VALLEY BOTTOM (including alluvial plain)	Major channel (river)	Mountain headwater	
					Mountain stream	
					Transitional	
					Upper foothill	
					Lower foothill	
					Lowland	
				Channelled valley bottom wetland	Channelled valley bottom depression	Exorheic
						Endorheic
					Channelled valley bottom flat	
				Unchannelled valley bottom wetland	Unchannelled valley bottom depression	Exorheic
						Endorheic
					Unchannelled valley bottom flat	
				Floodplain wetland	Meander cut-off	
					Backwater depression	
					Floodplain flat	
				Depression		Exorheic
						Endorheic
					Dam	
					Minor channel	
			SLOPE	Valleyhead wetland		
				Hillslope wetland	Hillslope seep	
					Hillslope depression	Exorheic
						Endorheic
					Minor channel	
			BENCH (HILLTOP / SADDLE / SHELF)	Valleyhead wetland		
				Depression		Exorheic
						Endorheic
				Flat		
					Minor channel	
			NON-ALLUVIAL PLAIN	Depression		Exorheic
						Endorheic
				Flat		
					Minor channel	

Each of the above levels of classification can be achieved using specific criteria that can be implemented in a GIS platform. To distinguish between marine and inland wetland systems in level 1 requires the use of a coastline boundary. But in South Africa there is no consolidated GIS database demarcating the coastline, as a result the land surface map is used to distinguish between marine and all inland wetlands. In level 2 the ecoregion setting or bioregional setting derived from vegetation setting (Musina & Rutherford 2006) is the basis for defining the regional setting. Level 3 of the automation framework, uses a topographic position index (TPI) to distinguish between various inland wetland types. The TPI uses data generated from a digital elevation model (DEM), to stratify the landscape based on the relative position of each pixel (**Weiss 2001**). In this way inland wetland systems can be classified on the basis of their position in the landscape. The level 3 wetland classification gives rise to four wetland types namely hilltop, slope, valley bottom and plain wetlands. The level 4 classification attempts to distinguish between wetland associated with surface flow, because water source plays an important role in determining the wetland function and benefits (**Nielsen et al. 2006**). The level 4 classification gives rise to five main wetland types (**Fig. 3.1**).

3.3 Applying the proposed protocol for wetland automation

3.3.1 Study Area

The approach discussed above was used to classify wetlands in the West Coast of the Cape Floristic Region, South Africa (**Fig. 3.2**). The study area is approximately 1158 km^2, and comprises part of the Oliphants river system, and its tributary the Doring, which are hotspots of fish endemism in Southern Africa. The topography of the region is highly diverse, ranging from the coastal lowlands in the east to the rugged mountain ranges in the West with an altitude of up to 2000m. The region experiences high summer rainfall (approximately 1500-2200mm), and mean temperatures of between -3 to 3 ^{o}C in winter, and between 39- 44 ^{o}C in summer. Most of the wetlands in the region are found in the lowlands, including some major estuarine systems such as the Langebaan estuary, which is a designated Ramsar site. The West Coast region was chosen for this study because of the diversity of wetlands occurring in this area. The major wetlands in the study area have also been mapped and classified at a fine scale, enabling them to be used as reference for testing the automation approach proposed.

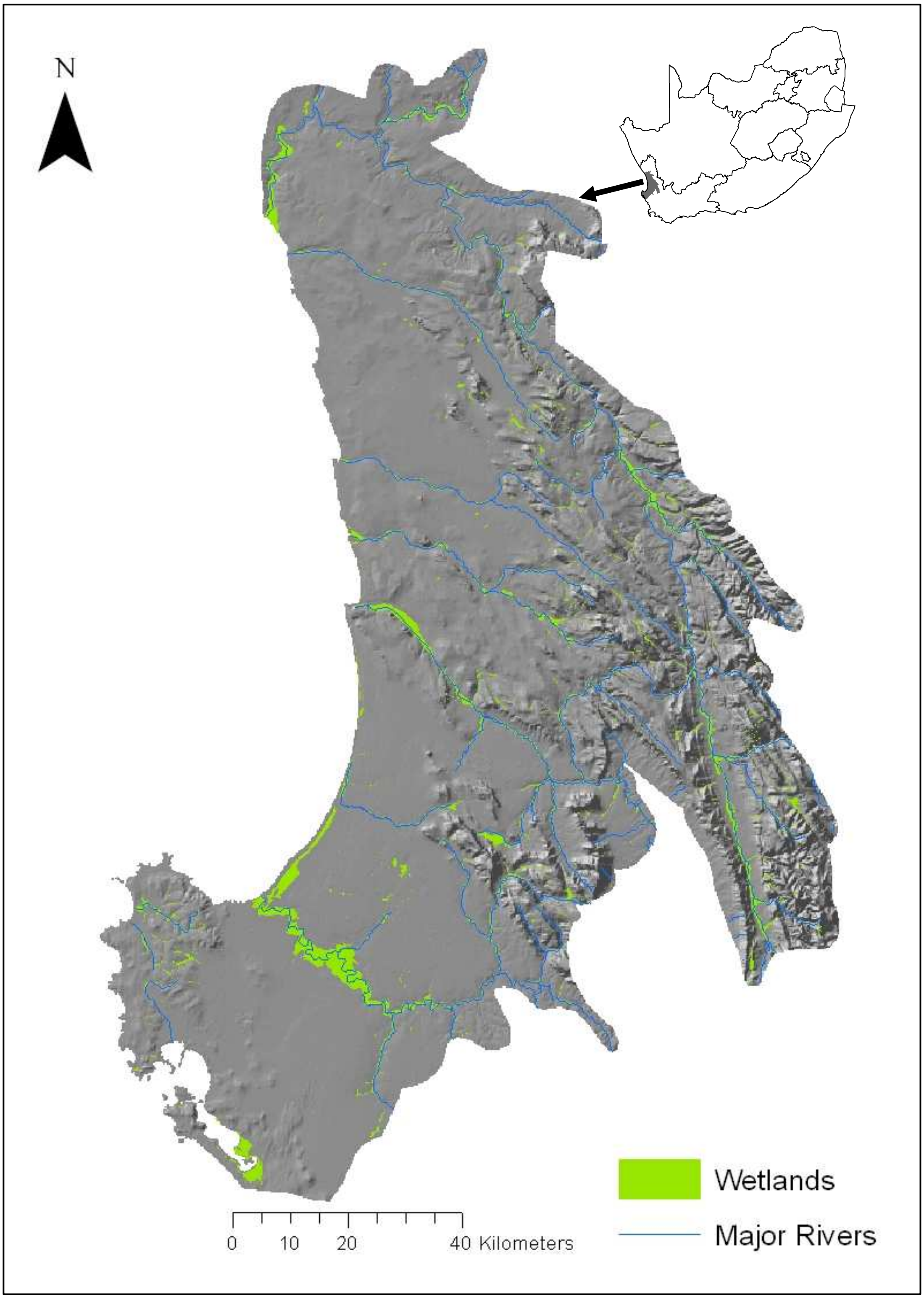

Figure 3. 2. Study area where the wetland automation framework was tested showing major wetlands and river systems in the region.

3.4 Data sources

Various datasets (**Table 3.2**) were used to implement the automation of wetland classification as described in the above criteria. The main data source for this project came from the National Wetlands Inventory (NWI). The NWI project aims to assess the extent, location and distribution of wetlands in South Africa. The aim of using the NWI dataset for this analysis was to enable the implementation of the automation approach for classifying the wetlands mapped by NWI nationally, to enable the assessment of national freshwater biodiversity priorities for South Africa. The second wetland dataset used to validate the wetland automation was derived from a major fine scale conservation plan for the Cape Floristic Region (CFR) that was conducted at a scale of 1:10, 000. The wetlands in this project were mapped with relatively high level of accuracy, and were classified based on the new wetland classification system for South Africa, whose automation is being pioneered in the current study. Other datasets used in the automation process included, a 90m digital elevation model (DEM) that was used to determine slope positions for distinguishing different wetland types based on landscape setting. Rivers mapped at a scale of 1: 50,000 were used to distinguish between riverine and non-riverine wetlands.

Table 3.2. Datasets there used to automate the wetland classification and to assess its effectiveness

Data	Description	Scale	Application	Source	Reference
National wetlands inventory map (NWI)	Wetlands mapped at national scale but not classified	1: 250,000	Used for testing the framework for automating wetland classification	CSIR-National wetland inventory project	http://bgis.sanbi.org/nwi/project.asp
DEM	This is a modified SRTM 90m digital elevation model	90m	Used to generate a topographic position index (TPI) and slope positions	CGIAR Consortium for Spatial Information	http://srtm.csi.cgiar.org
Cape fine scale plan wetland map (FSP)	Wetlands that were mapped and classified at a fine scale for the Cape Fine Scale conservation planning project	1:10,000	Used to compare the accuracy of the automation framework for wetland classification	South African National Biodiversity Institute (SANBI)	www.capeaction.org.za
Rivers shapefile	Rivers that were mapped at a national scale showing river orders 1- 7	1:50,000	Used to distinguished between different wetland types associated with rivers	Dept. of Land Affairs	DLA 2005
Buffered rivers	100m buffer on main stem rivers	1:50,000	Used to distinguished between different wetland types	N/A	DLA 2005

3.5 Wetland classification

3.5.1 Level 1 & 2 classification: Marine and inland wetlands

In this study the wetlands that were being classified were from a database that comprised of inland wetlands only, as a result automating level 1 of the classification was not necessary. Level 2 of the classification system was also not implemented while testing the automation framework, because it is argued that specifying the regional setting before the landscape setting as defined in the wetland classification framework (**Table 3.1**) complicates the automation process. The regional setting results in numerous variations in wetland types before applying the criteria based on landscape setting, which is the key geomorphic criterion for distinguishing the most basic wetland types (i.e. valley bottoms, slope, plain and hilltops).

3.5.2 Level 3 classification: Major inland wetland types

A topographic position index (TPI) was generated using a 90m DEM, where the algorithm compared the relative position of each cell in the DEM to the mean elevation of a specified neighbourhood cell (**Weiss 2001, Judex et al. 2006**). Relative position was defined as the difference in the height value of a location from surrounding mean height values (**Judex et al. 2006**). If the TPI value of a specific location is positive, it means that location is at a higher elevation than the surrounding neighbourhood and therefore a ridge, while negative TPI value represents locations with a lower elevation than their surrounding neighbourhood and therefore a valley. TPI values close to zero represent flat areas, or areas with a constant slope (**Weiss 2001**).

Slope positions of the entire study area were calculated by converting TPI values into thresholds using the standard deviation (SD) of their elevation and slope at the specified location. Using the SD of elevation instead of direct TPI values enabled the variation in the landscape to be taken into account when calculating slope positions (Weiss 2001). This was important because, for example, no two valleys occurring in different regions are the same. The SD factors in the elevation of the neighbouring cells, as a result depending on the region of analysis, two separate locations having the same TPI values might not necessarily be classified as the same due to the variation in the elevation of their neighbourhood cells. Based on this approach, the landscape was classified into six main categories: ridges, upper slope, middle slope, flat slope, lower slope and valley. To make this classification relevant to the wetland classification framework discussed earlier, the above categories were regrouped into the following categories using the standard deviation thresholds stated:-

- Hilltop [TPI > 1 SD]
- Slope [TPI > = -1 SD and <= 1 SD]
- Plain [TPI >= -0.5 SD and <= 0.5 SD]
- Valley Bottom [TPI < -1 SD]

TPI values are scale-dependent because of neighbourhood size, whereby small neighbourhoods capture small and local hills and valleys, while large neighbourhoods capture larger-scale features (**Jenness 2006**). To decide on the most appropriate neighbourhood size depends on the landscape and the objective of the study. In this analysis five neighbourhood cell sizes of 10, 20, 40, 60, 80, and 100 with a cell size of 30m each, were assessed based on a trial and error basis, to determine their accuracy in predicting the wetland type. Once the correct neighbourhood size was determined, the TPI was generated and later used with the slope data to stratify the entire landscape into a slope position, giving rise to the four categories mentioned above. To classify the wetlands into different types, the unclassified wetland layer from the NWI was intersected with the slope position as determined by the TPI. Accuracy of classification was determined by comparing wetlands that have been classified in this way with a separate group of wetlands that were mapped and classified at a fine scale.

3.5.3 Level 4 classification: Wetlands associated with river channels

For level 4 classification, valley bottom and slope wetlands were further classified based on their association with rivers. The 1:50,000 river coverage for South Africa was used to determine slope wetlands associated with 1st and 2nd order rivers, to classify slope wetlands into hillslope seeps and hillslope depressions. Valley bottom wetlands associated with 3rd or higher order river channels, were used to classify them into channelled and unchannelled valley bottom or floodplain wetlands. A buffer strip of 100m on both sides of lowland rivers was used to distinguish between floodplain and unchannelled valley bottoms. The accuracy in the classification was determined as the percentage of the grid cells from the unclassified wetland dataset that were assigned the correct wetland class, relative to the wetlands that were mapped and classified at a fine scale.

3.6 Results

3.6.1 Effect of scale on wetland mapping

The scale at which the wetlands were mapped had an effect on the accuracy of wetland delineations. In total 1406 individual wetlands were mapped at the fine scale, but the broad scale wetlands mapped by the NWI, recorded only 17% of the total area of wetlands (**Fig. 3.3**). Although the broad-scale mapping grossly underestimated the area of wetlands, observations suggest that most of the individual wetlands were identified even though their boundaries were not accurately mapped. According to the fine scale wetland map, the majority of wetlands in the region were valley bottom wetlands (83%), followed by plain wetlands (15%), with only about 1% of slope wetlands.

3.6.2 Level 3 wetland classification

Sensitivity analysis of the topographic position index (TPI), based on varying neighbourhood cell sizes showed negligible variation in the accuracy of wetland classification within the same datasets. For the NWI wetlands a neighbourhood cell size of 100m was more accurate in classifying the wetland types, but the fine scale wetlands a neighbour cell size of 60m was more accurate when the fine scale wetlands were classified using the proposed approach. The level 3 wetland classification produced four main wetland types namely valley bottom wetlands, plain, slope and hilltop wetlands (**Fig. 3.4**). Both the wetlands mapped at a broad scale and those mapped at fine scale showed relatively comparable level of accuracy (**Table 3.3**).

3.6.2.1. Valley Bottom wetlands

For both the broad and fine scale wetland maps, valley bottom wetlands were the largest group of wetland types in the study area encompassing more than 80% of the total area of wetlands. The broad scale wetlands classified as valley bottom had an average area of 1.6ha. The accuracy of classification of valley bottom wetlands was also highest among all the other wetland types, at 89% for the broad scale wetlands, and 73% for the fine scale wetlands. For the broad scale NWI wetlands, about 10% were a misclassification of plain wetlands and 0.7% of slope wetlands (**Table 3.4**).

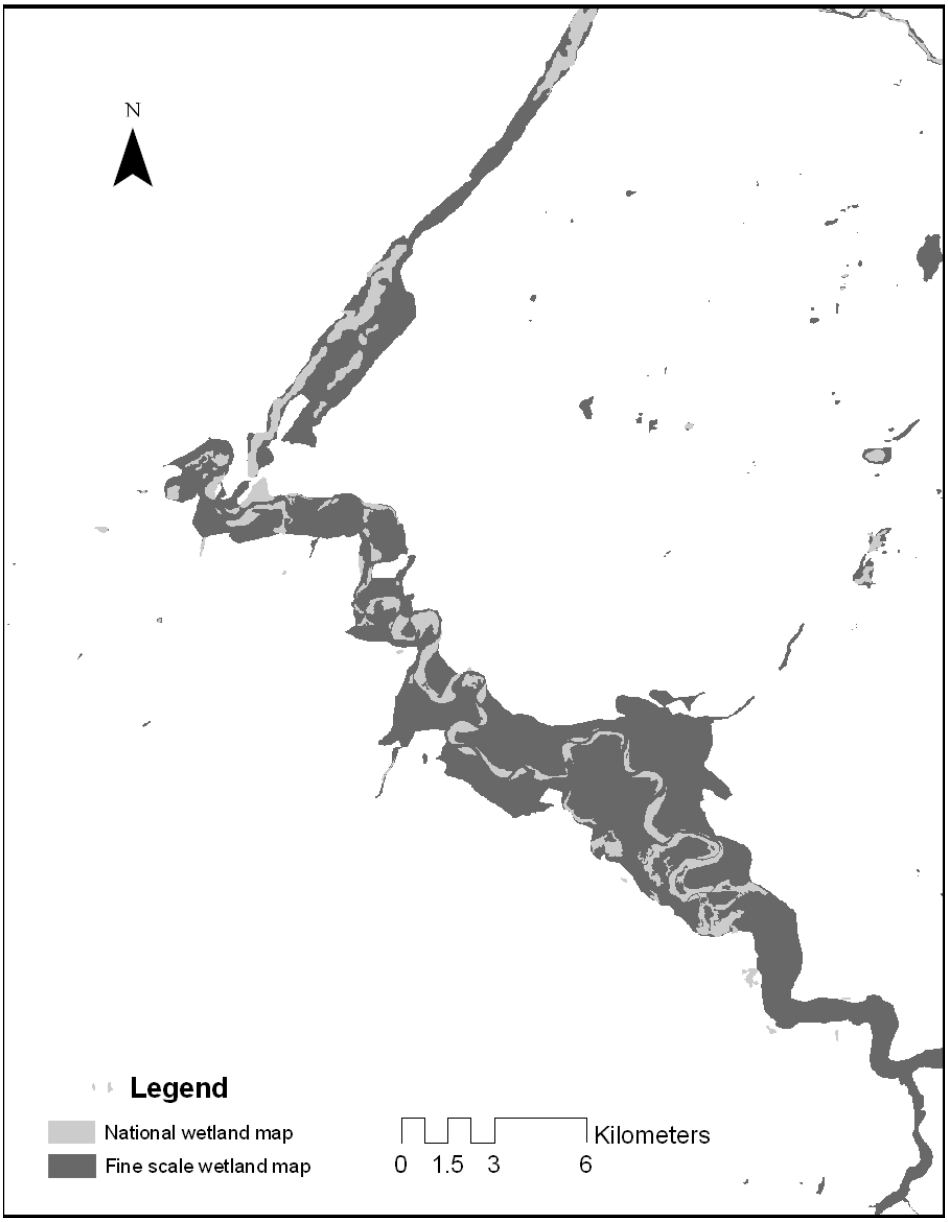

Figure 3.3. Illustration of the disparity in mapping wetlands at a broad scale by the National Wetland Inventory (NWI) project, which only captured 17% of the total area of wetlands as mapped at a fine scale.

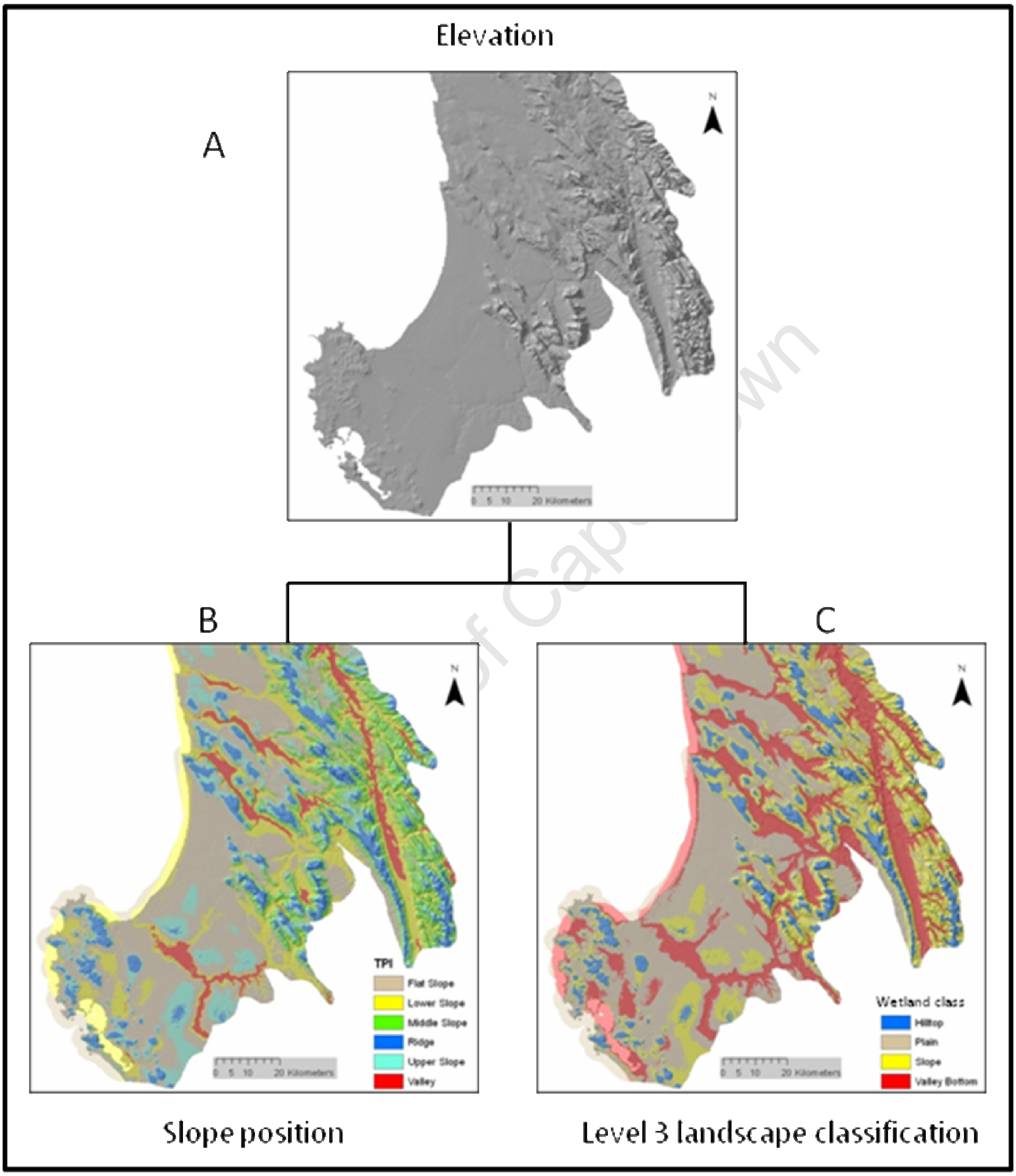

Figure 3.4. A schematic illustration of the process of classifying wetlands at level 3 of the classification framework. Map A is the elevation, which was used to derive a topographic position index (TPI) for each grid cell in the landscape. Slope was also calculated from the elevation model and used in conjunction with the TPI to derive a slope position with 5 classes (Map B). The slope position was then reclassified to correspond with the wetland classes suggested in level 3 of the classification framework (Map C).

Table 3.3. Level 3 of the wetland classification showing the application of the automation framework for classifying wetlands mapped at broad and fine scale. Accuracy of the classification was based on how the classification achieved from the automation framework compares with that of the fine scale wetlands that were mapped accurately in the field.

All NWI wetlands	**Average area (ha)**	**Total area (ha)**	**% of each wetland type**	**% area of wetlands mapped at broad scale**	**Accuracy of classification (%)**
Hilltop	0.2	4.4	0.6	n/a	0.0
Plain	0.8	100.3	14.6	n/a	50.3
Slope	0.4	16.1	2.3	n/a	74.0
Valley Bottom	1.6	568.5	82.5	n/a	89.0
ALL fine scale wetlands	**Average area (ha)**	**Total area (ha)**	**% of each wetland type**	**% area of wetlands mapped at broad scale**	**Accuracy of classification (%)**
Hilltop	0.0	0.0	0.0	0.0	0.0
Plain	1.9	235.3	5.8	42.6	45.0
Slope	2.3	379.5	9.4	4.3	49.5
Valley Bottom	4.2	3418.0	84.8	16.6	73.1

Table 3.4. A comparison of how the wetlands mapped at national scale based on the NWI were classified relative to the fine scale wetland types.

		Wetlands from the National Wetland Inventory (NWI)				
		Hilltop*	Plain	Slope	Valley bottom	Total
Fine scale wetlands	Slope	n/a	31(3.5%)	26(74.3%)	2 (0.7%)	53
	Plain	n/a	451(50.3%)	7(20%)	275(10.1%)	733
	Valley bottom	n/a	414 (46.2%)	2(5.7%)	2421(89.1%)	2861
	Total	n/a	896	35	2716	3647

*None of the fine scale wetlands were classified as hilltops so comparison was not possible

3.6.2.2. Plain wetlands

Plain wetland types in relation to landscape setting are those wetlands in relatively flat areas and not surrounded by a slope in both sides (**Fig. 3.5**). Plain wetlands from the broad scale NWI wetland map comprised about 15% of the total area of wetlands in the study area, and were of an average size of 0.8ha. They were classified with an accuracy of about 50.3%. About of 46.2% of wetlands classified as plain were a misclassification of valley bottom wetlands and 3.5% were slope wetlands according to the classification of wetlands at the fine scale (**Table 3.4**). For the wetlands mapped at a fine scale, when the proposed approach in this study was used to classify them, an accuracy of 45% was achieved.

3.6.2.3. Slope wetlands

Slope wetlands are those that occur in areas with a very steep gradient, for example wetlands found on the side of a mountain or a hill. About 0.2% of the total area of wetlands in the study area was classified as slope wetlands at the broad scale. The accuracy estimation found that 74% were accurately classified relative to the wetlands that were mapped at a fine scale. The remaining 20% and 5.7% were a misclassification of valley bottom wetlands and plain wetlands respectively (**Table 3.4**). The fine scale wetlands achieved an accuracy of 50% when they were classified using the same approach as the broad scale wetlands.

3.6.2.4. Hilltop wetlands

Hilltop wetlands occur on relatively flat areas at a high elevation, and could be surrounded on both sides by a steep gradient. In the classification, 0.6% of the total area of the NWI wetlands was classified as hilltop wetlands. However, the wetlands that were used as reference for the classification did not have any hilltop wetlands. This appears to have been an oversight in the classification approach adopted for fine scale wetlands that were used as a reference for this study.

3.6.3 Level 4 wetland classification

The level 4 classification attempted to introduce hydrological aspects by classifying wetlands based on their association with flowing surface waters. This classification gave rise to five wetland types with very low degree of accuracy for the broad scale wetlands. In comparison to level 3 of the classification hierarchy there was a high level of uncertainty in classifying the wetlands mapped at the broad scale from the NWI database. The results showed that only channelled valley bottom wetlands were accurately classified (**Table 3.5**). All the unchannelled valley bottoms were mis-classified either as channelled valley bottom (91%), lowland floodplain (7%) or hillslope seep (3%). The accuracy of hillslope seep wetlands ranged between 10.4%, while lowland floodplains were poorly predicted too, with a range of between 1.6% accuracy. When the wetlands mapped at fine scale were classified using the proposed approach, the accuracy in their classification appeared to be more consistent than for the broad scale wetlands (**Table 3.5**).

Table 3.5. Level 4 of the wetland classification framework was implemented for both the NWI wetlands and fine scale wetlands. At this level Hilltop and Plain wetlands could not be further classified, and were excluded from the classification. The NWI wetlands were very poorly classified compared to the fine scale wetlands.

All NWI wetlands*	**Average area (ha)**	**Total area (ha)**	**% of each wetland type**	**% area of wetlands mapped at broad scale**	**Accuracy of classification (%)**
Hillslope Seep	0.04	12.0	1.9	n/a	10.4
Hillslope depression	0.20	0.2	0.0	n/a	0.0
Channelled Valley Bottom	3.93	469.8	73.1	n/a	62.4
Unchannelled Valley Bottom	4.49	78.0	12.1	n/a	0.0
Lowland Floodplain	0.22	83.1	12.9	n/a	1.6

ALL fine scale wetlands*	**Average area**	**Total area (ha)**	**% of each wetland type**	**% area of wetlands mapped at broad scale**	**Accuracy of classification (%)**
Hillslope Seep	1.6	379.4	10.0	3.2	n/a
Hillslope depression	0.0	0.0	0.0	0.0	0.0
Channelled Valley Bottom	17.6	2045.4	53.9	23.0	53.8
Unchannelled Valley Bottom	3.0	170.8	4.5	45.6	20.9
Lowland Floodplain	34.6	1201.8	31.6	6.9	28.0

* excludes hilltop and plain wetlands that cannot be classified further

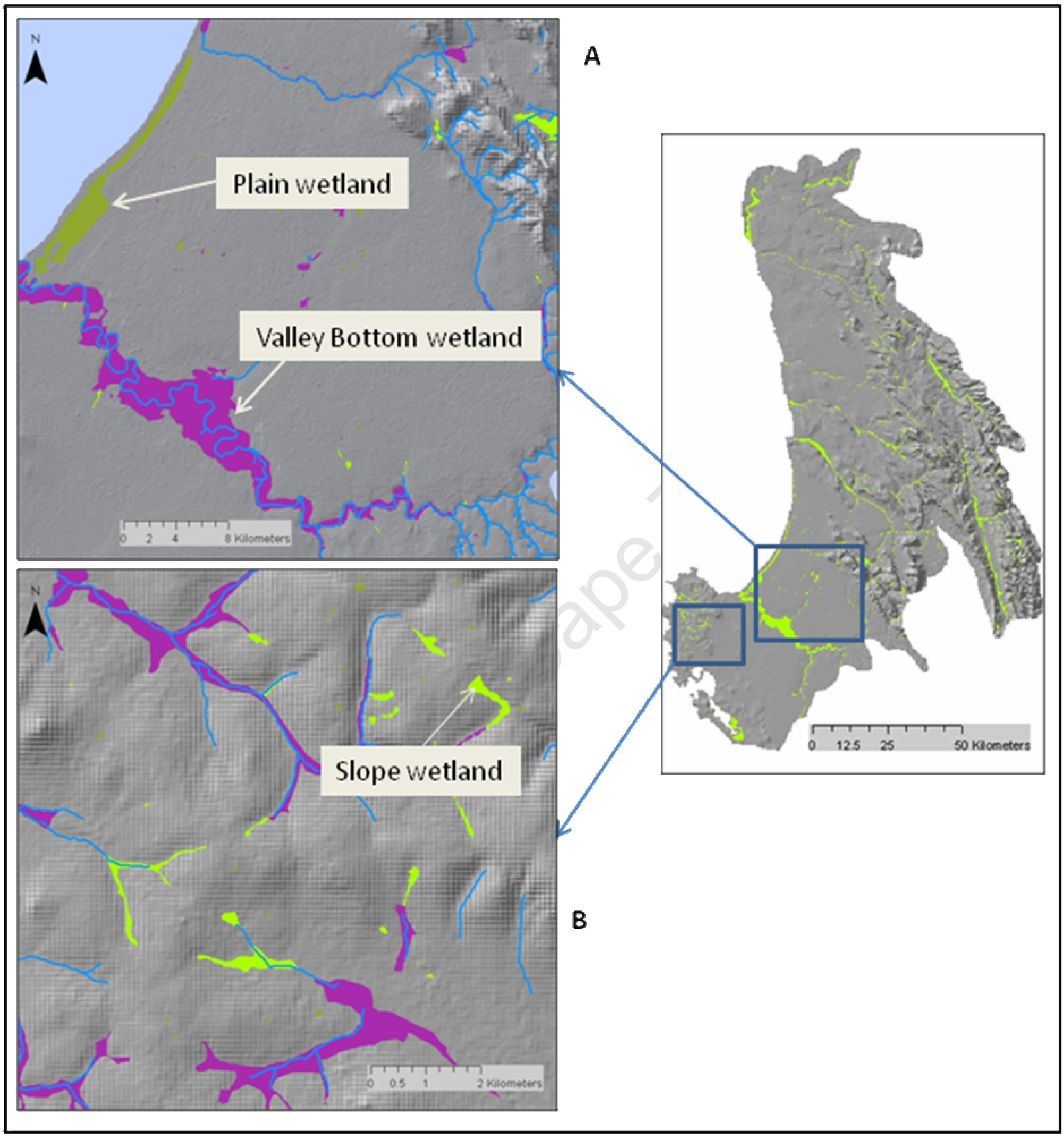

Figure 3.5. An example of the different wetlands classified at level 3, and the major rivers associated with some of the wetlands.

3.7 Discussion

Incorporating wetlands comprehensively in conservation planning still remains a major challenge (Chapter 2, Nel et al. 2008). This study aimed to devise ways of effectively incorporating wetlands into conservation planning frameworks, by ensuring that they can be rapidly classified in order to capture the inherent diversity within wetland ecosystems. This was achieved through the application of a GIS-based approach for automating wetland classification as mapped by the national wetland inventory project. One of the key insights gained in this study was the fact that wetland classification is a complex process, but a prerequisite for wetland conservation (Finlayson & van der Valk 1995, Semenuik & Semenuik 1997). The level of detail into which wetland classification is undertaken should be determined to a large extent by the objectives of the wetland classification, and data availability (Smith et al. 1995, Scott & Jones 1995). Patchy wetland data is a major constraint in South Africa, it is therefore important that the wetland classification system is designed to take this issue into consideration.

The proposed criteria are a simple approach for the rapid automation of wetland classification, based on their geomorphic setting. The framework provides a step-by-step approach to classifying wetlands, in a way that can be easily applied by anyone with the most basic GIS and wetland expertise. The use of spatially explicit map-scale data as the main discriminators of wetland types enabled the classification to be based on free and widely available datasets consistent with Williams et al. 2000. This is important because the lack of data on wetlands is a major challenge both locally and globally. For this reason the implementation of effective wetland conservation strategies has been very challenging. The prevalence of aerial photographs and other remotely sensed images is set to leverage this predicament, enabling a landscape scale approach to solving ecological problems. The need for the automation of these processes at a landscape scale is inevitable as they tend to be time consuming and labour intensive (Williams et al. 2000). The approach proposed in this study is a useful one that will facilitate wetland assessment for conservation planning, because it is comprehensive for classifying wetlands, a key requirement for conservation planning (Roberson & Fitzsimons 2004).

The approach was found to be relatively useful in classifying wetlands up to level 3 of the proposed wetland classification framework for South Africa (Table 3.1). The four wetland classes (hilltop, slope, plain and valley bottoms) provided a basic classification of wetlands that was broad enough to incorporate all inland wetland types including estuaries. This was important because of the variation in regional setting and geomorphology in different parts of the country. This framework therefore lends itself to application in other regions, where it might not have been tested. The four wetland categories were delineated based only on geomorphology, which enabled the basic structure underlying wetlands to be highlighted (Semenuik & Semenuik 1997), and the next level of the classification incorporated the hydrological aspect of wetlands. This logical approach enables wetland classification to be driven primarily by the objectives of the classification (Smith et al. 1995), scale of analysis and data availability. In cases where a fine scale wetland classification is required then the hydrological aspect to wetland classification could be introduced leading to the implementation of level 4 in the hierarchy, but for national scale conservation planning purposes a level 3 wetland classification is adequate.

Level 4 in the classification hierarchy is more suitable for wetlands mapped at a fine scale, because of the difficulty in specifying the water source for individual wetlands. In this analysis, the proximity of wetlands to rivers was used as a surrogate to define the water source for each wetland system. But accuracy of the wetland classification was very low for the wetlands mapped at the broad scale (Table 3.5). The failure of this approach to classify wetlands based on their association with flowing surface waters could be attributed to several factors.

Firstly, the use of river orders and typology appears to be ineffective for determining different slope and valley bottom wetland types. Whether this anomaly was a result of the scale of analyses, DEM resolution or the 1:50,000 river layer, it could not be ascertained. Secondly a lack of clear definition between unchannelled valley bottom wetlands and lowland floodplain could have contributed to the misclassification. Both wetland types occur in lowland terrain and because elevation plays a key role in determining the wetland types it became difficult to distinguish between the floodplain wetlands and the unchannelled valley bottoms. Thirdly the use of arbitrary discriminators such as buffer distance to distinguish between lowland floodplains and unchannelled valley bottoms may have contributed to the inaccuracy in classification. Finally, any errors in the DEM used to generate the topographic position index (TPI), could also lead to misclassification of the valley bottom wetlands (Williams et al. 2000).

Even though most of the wetland types in level 4 were not accurately classified, most of the wetlands were still nested within the same class at the broader level 3. For example, lowland floodplains and unchannelled valley bottoms were classified with very low accuracy (Table 3.5), but more than 90% of the misclassification were attributed to channelled valley bottom wetlands. This implies that the misclassification is not too gross as both unchannelled valley bottoms and the lowland floodplains are essentially valley bottom wetlands in reference to level 3 in the classification hierarchy. The misclassification of wetlands at level 4 of the classification hierarchy does not therefore have major implications when such an approach is used to derive wetland types for representation in conservation planning.

The essence of wetland classification is to develop appropriate surrogates for biodiversity, and although it has been recognised that different wetland types have unique functions and meet different species requirements (Roe & Georges 2007), there is a need to test the efficacy of wetland classification in representing biological diversity. This is an important exercise that was not achieved in this study, because it was beyond its scope of aiming to automate wetland classification. It would be useful to compare for example, the outcome of a conservation plan based on the wetland classes suggested in this study and one based on species data to determine whether the same set of priority areas will be identified. Alternatively a simple cluster analysis of species richness and or assemblages based on the suggested wetland types would be a useful indicator of their effectiveness in representing wetland biodiversity.

The fact that the national wetland inventory (NWI) only captured 17% of the total area of wetlands in the study area is a poor attempt to comprehensively map wetlands at a national scale. The importance of having a comprehensive national wetland database is a prerequisite for the identification of the network of potential conservation priority areas to meet South Africa's national and international obligations. For example the convention on biological diversity (CBD) and the Ramsar convention, of which South Africa is a signatory stipulates that member countries meet specific obligations, such as the listing of wetlands of international importance and the development of a national biodiversity action plan (NBSAP). Nationally the South African Biodiversity Act (2004) also stipulates that threatened ecosystems should be listed, and that process has largely been accomplished for terrestrial ecosystems. For rivers, the potential of listing the last free flowing rivers in the country as Heritage Rivers is currently being explored. No progress has however been achieved in listing threatened wetland ecosystems, because no comprehensive national wetland database exists, and the proposed wetland classification systems has not been implemented at a national scale. This study is therefore of significance in the context of wetland conservation in South Africa, because it is the first attempt to automate the national wetland inventory (NWI). Even though it might not have been entirely successful, it has set the scene for further work in ensuring that important wetland systems can be identified nationally in a manner that is comprehensive, adequate, and representative of the wetland ecosystems in South Africa.

CHAPTER 4. PLANNING FOR FRESHWATER BIODIVERSITY PERSISTENCE IN THE CAPE FLORISTIC REGION, SOUTH AFRICA

ABSTRACT

The Cape Floristic Region (CFR), situated in South Africa is a global biodiversity hotspot, well known for its high level of plan diversity and endemism. It is home to more than nine thousand vascular plant species, with more than 60% endemic to the region. In terms of freshwater biodiversity it is less diverse compared to the terrestrial ecosystem; however it also exhibits a very high level of endemism, with 16 out of 19 fish species endemic to the region. The conservation of freshwater biodiversity in the CFR has lagged behind terrestrial ecosystems, because most protected areas in the region were designed for terrestrial biodiversity. A systematic freshwater conservation plan was carried out for the region to identify the gaps prevalent in the protection of freshwater biodiversity, and to demonstrate a holistic approach to freshwater conservation planning for biodiversity persistence. The results showed that in order to secure adequate protection of freshwater ecosystems in the CFR, at least 30% of the area should be set aside for conservation in addition to the current network of protected areas. It was also found that lowland river systems were most impacted by anthropogenic disturbances, and contributed 8% towards achieving conservation targets. There is thus an urgent need to redesign the current network of protected areas in the region to meet freshswater conservation goals. However, this may involve a trade-off between freshwater and terrestrial biodiversity priority areas, as they do not often overlap.

4.1 Introduction

The main goal of systematic conservation planning is to identify, design and implement a network of conservation priority areas that are representative of the region's biodiversity, and will result in their persistence (Williams & Araujo 2000, Margules & Pressey 2000). For long time terrestrial conservation planning techniques and methods were focused on achieving representation (Knight et al. 2006, Strange et al. 2007), therefore optimising for representation and to a lesser extent persistence of terrestrial biodiversity is relatively well understood. In freshwater ecosystems however, these are still pertinent issues, it was only recently that freshwater conservation planning issues started to take centre stage (see Abell 2002, Nel et al. 2008). Progress in freshwater conservation planning is however, still lagging far behind terrestrial and marine ecosystems (Chapter 2).

The many facets of freshwater ecosystems have particularly made it challenging to incorporate all the aspects ranging from estuaries, wetland and rivers into a single conservation assessment framework. Wetlands for example present enormous challenges in conservation planning because they occur in transition zones between terrestrial and freshwater ecosystems, so in many conservation assessments wetlands have been considered as part of the terrestrial landscape (e.g. Lombard et al. 1997, Madsen & Clausen 1998). The incorporation of wetlands as a terrestrial vegetation type could arguably be regarded as the most effective means of protecting wetlands so far (Rouget *pers. Comm.*). However in most cases wetlands are incorporated into terrestrial conservation planning frameworks only because of their importance as habitat for terrestrial species such as migratory bird species.

Effective wetland conservation requires the classification of wetlands into their different types, a practice that is seldom applied in systematic conservation planning. This has been compounded by the fact that most wetland classification frameworks are theoretical and difficult to apply in a conservation planning environment (see Chapter four). A related issue but one that affects both wetlands and rivers, is that of incorporating freshwater ecological integrity in conservation planning. For biodiversity persistence, priority ecosystems must have a high ecological integrity so that only rivers and wetlands with a high potential to ensure biodiversity persistence are listed as priority conservation areas (Nel et al. 2008). Other requirements for biodiversity persistence include maintaining upstream-downstream connectivity for rivers and protecting vital ecological processes (Linke et al. 2008). A major shortfall in the current freshwater conservation planning frameworks is the inconsistence with which principles of biodiversity representation and persistence are applied when identifying conservation priority areas.

In Chapter Two, the major constraints to effective freshwater conservation planning were highlighted, which included the challenges in integrating freshwater and terrestrial priority areas, assessing freshwater ecological integrity and incorporating wetlands into conservation planning frameworks. In this Chapter, the Cape Floristic Region (CFR) is presented as a case study of how the issues raised in the previous chapters can be incorporated in a typical systematic conservation planning scenario. The conservation planning approach undertaken in the CFR sought to specifically address the following issues:-

- The effective incorporation of wetlands into freshwater conservation planning approaches
- Accounting for freshwater biodiversity persistence in conservation planning

Both of these issues are not new to conservation planning, but this is the first time a comprehensive freshwater conservation planning is being undertaken for the CFR that particularly seeks to address these challenges.

The CFR is one of 35 global biodiversity hotspots due to the uniqueness of its biodiversity (Meyers et al. 2000). Freshwater ecosystems in the Cape Floristic Region exhibit a very high level of species endemism, although less diverse compared to terrestrial ecosystems. Fish species are the most endemic biota in the CFR, with 16 out of 19 species endemic to the region (Impson & Cambray 2002). The high level of endemism in freshwater biodiversity could be attributed to the unique Mediterranean climate and acidic waters to which many species have adapted (Thieme et al. 2005). Wetlands form an important component of the CFR, but are by far the least studied of freshwater ecosystems in the region and South Africa at large (Grenfell et al. 2005).

Freshwater ecosystems in the CFR are highly threatened by land use change, and is predicted to be highly impacted by climate change (Williams et al. 2005). More than 20% of the area has been cultivated, and thus agriculture and its associated activities are one of the major threats to freshwater ecosystems in the region (Cowling 2003). Alien invasive fish species are a major threat to the indigenous fish, having being introduced in many river systems for recreational and other purposes in the past. Although many river systems remain free of alien fish, in some streams up to eleven alien fish species have been recorded (Thieme et al. 2005). Alien fish species negatively impact on indigenous species through competition and predation (Van Nieuwenhuizen 2000), and sometimes they can temporarily change the physical characteristics of the river system by stirring up sediment and increasing turbidity (Van Nieuwenhuizen 2000). The most common invasive fish species in the CFR are bass (*Microoterus* spp) and trout (*Oncorhynchus mykiss and Salmo trutta*), which have reportedly eliminated several indigenous species of the genus *Pseudobarbus* from parts of their ranges (Van Nieuwenhuizen 2000).

More than 10% of the region is under some form of protection, and most are in the IUCN categories I to IV (WDPA 2004). The protected areas in the CFR are however, mostly located in mountainous regions with very few in the lowlands (Rouget et al. 2003); as a result some of the most critically endangered freshwater systems are the middle and lower river reaches (Impson & Cambray 2002). Most of the protected areas were not designed with the intention of conserving freshwater ecosystems, a trend that has been observed in South Africa (Roux et al. 2008), and other parts of the world (Mancini et al. 2000, Keith 2000). The CFR is no exception, for example the most recent systematic conservation plan for the CFR (Cowling et al. 2003) focused mostly on terrestrial ecosystems with little consideration of freshwater biodiversity in the assessment.

There is therefore a need for a comprehensive freshwater conservation plan for the CFR to identify priorities for conserving freshwater biodiversity, evaluate the extent to which current conservation efforts are achieving freshwater biodiversity goals and to chart the way forward. This study therefore aims to develop a comprehensive freshwater conservation plan for the CFR that incorporates effectively all the different freshwater ecosystem types, address challenges of assessing their ecological integrity, and evaluate the extent to which current conservation efforts are achieving freshwater conservation goals in the CFR.

4.2 Methods

4.2.1 Region of analysis

South Africa has been divided into 19 water management areas (WMAs) for management at catchment level. In the CFR there are four WMAs: Oliphants/Doring, Berg, Breede and the Gouritz. This study comprised the last three WMAs, with a total area of 78933Km² (Fig. 4.1), the Oliphants/Doring WMA, which contains the highest number of endemic fish species in South Africa, was not included in this assessment because a conservation plan has been developed for that region (Nel et al. 2006).

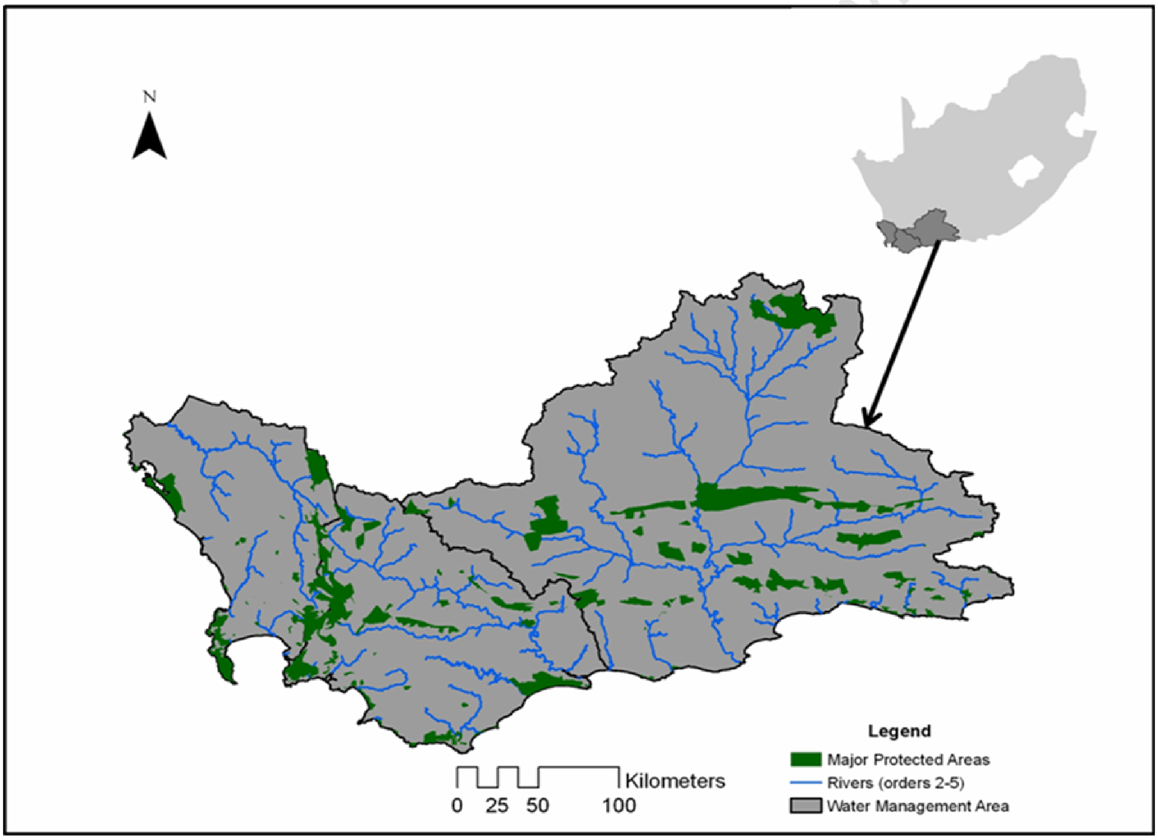

Figure 4.1. The study area in the Cape Floristic showing the three water management areas, major river systems and protected areas (category I & II)

Most parts of the Cape Floristic Region have a Mediterranean type of climate (Davies et al. 1995), characterised by winter rainfall, with annual rainfall ranging from 600 to 2000mm. Many of the rivers in the Fynbos ecoregion are peat-stained, acidic and generally nutrient-poor (Van Nieuwenhuizen 2000), while rivers in the east of the region are seasonal or ephemeral, and alkaline. Rivers in the south coast are similar to those found in the fynbos ecoregion, they are clear, peat-stained and often alkaline, but are generally shorter than 20km (Van Nieuwenhuizen 2000). Wetlands are an important component of freshwater ecosystems in the CFR. They are quite diverse and their distribution is determined mainly by geomorphology and climate (Ewart-Smith et al. 2006). The flat and low lying areas favour the formation of pans, whereas the moist areas favour the formation of perennial endorheic wetlands. The foothills favour the formation of wetlands along streams and seeps, while the coastal areas favour the formation of estuarine lagoons and salt marshes (Van Nieuwenhuizen 2000).

The steps followed to undertake freshwater conservation planning

Systematic conservation planning was undertaken in several stages, based on well established principles (Margules & Pressey 2000), but also taking into cognisance the uniqueness of freshwater ecosystems.

1) Identify the planning domain where systematic conservation planning is to be undertaken
2) Set conservation goals
3) Compile data on biodiversity pattern and processes in the planning domain
4) Formulate biodiversity targets
5) Assess the ecological integrity of the freshwater ecosystems
6) Run the planning tool (MARXAN) to identify the biodiversity focal areas
7) Build in connectivity of the focal areas
8) Review target achievement by the potential conservation priority areas and the current network of protected areas
9) Assess the overlap between freshwater and terrestrial priority areas
10) Expert review of potential network of conservation priority areas

4.3 Mapping biodiversity pattern and processes

4.3.1 Physical river types

River types are one of the most widely used surrogates for representing riverine biodiversity in conservation planning (e.g. Roux et al. 2008, Thieme et al. 2007). Rivers are typed in a broad physical classification based on a variety of factors such as flow and geomorphology (Table 4.1). Broad surrogates such as the river types aim to conserve biodiversity at a higher level of the biodiversity hierarchy such as communities, and populations (Thieme et al. 2007, Roux et al. 2008). The use of such a classification is based on the premise that diversity in the physical characteristics of a river system plays an important role in determining the type of habitat available for riverine organisms. The rate of sediment transport, is faster for example in mountain streams than in lowland rivers, and the rate of sediment deposition in the river bed will determine the type of habitat instream species can occupy. The aim of delineating river types is therefore to capture as much diversity in the river systems as possible.

Three main physical templates were used to delineate river types in this assessment. These were river ecoregions (Kleynhans et al. 2005), hydrological flow types (Hannart & Hughes 2003), and geomorphological river zones (Rowntree & Wateson, 1999). Ecoregions are a hierarchical representation of the landscape based on the similarity of their biotic and abiotic composition (Omernik 1987). In South Africa, rivers have been classified into ecoregions, which has given rise to 31 level I ecoregions in the entire country, and seven of those occur in the study area. For this study we used the finer-scale ecoregion level II classification with ten level II ecoregions represented in the study area. The second variable used to classify rivers is derived from a hydrological index (Hannart & Hughes 2003), which generated a single statistic reflecting temporal variability of river flow (Roux et al. 2008) and classifying rivers as perennial, seasonal and ephemeral rivers. The third input to the river classification was a geomorphological index (Rowntree & Wadeson1999), which stratified the rivers into four longitudinal zones: (i) mountain stream, (ii) upper foothill, (iii) lower foothill and (iv) lowland rivers. To generate the river types, information on the ecoregion classification, flow variability and the geomorphological river zones were overlaid in a GIS (Fig. 4.2).

Table 4.1. Summary of biodiversity features used to identify freshwater priority areas in the CFR

Biodiversity Feature	Description	Extent/Size	Target
Rivers – biodiversity	Rivers classified based on geomorphology, hydrology and Ecoregion Level 2 as surrogate for biodiversity	15604.42 km of rivers, representing 165 types, stream orders >2	20% flat target in line with the national spatial biodiversity assessments (NSBA)
Seepage and valley bottom wetlands	Wetlands classified into functional types, then according to ecoregion level 2 biodiversity template	Only wetlands 0.2 ha or greater; a total of 19504 ha (22% of all wetlands types)	A baseline target of 15% and adjusted upwards according to biodiversity value
Floodplain wetlands		35285.31 ha	24%
Pan wetlands	Scale of 1: 50 000. Classified into perennial and non-perennial pans, and ecoregion level 2 biodiversity template	3197.34 ha of pans	Base target of 24%, adjusted upwards according to biodiversity value
Important wetland clusters	Buffer all wetlands greater than 5 ha with 1 km and remove transformed areas. Include clusters 1 000 ha or greater, with area to perimeter ratio > 300	1198 ha.	Base target of 24% for all important wetland and pan clusters
Estuaries		23322.69 ha of estuaries	20%of target
Lagoons & other unclassified wetlands		6069.33 ha	Base target of 20%
Fish species	Known distribution of 4 threatened fish species	4 fish modeling known distribution; 908 km of river length	Target based on rarity and distribution, ranged from 50- 100%
Fish Sanctuaries		235381.16	
High water yield areas	Median annual runoff per quaternary catchment. Select catchments producing 50% of runoff as high water yield areas	High water yield areas equate to 15% of the planning domain	Base target of 50%

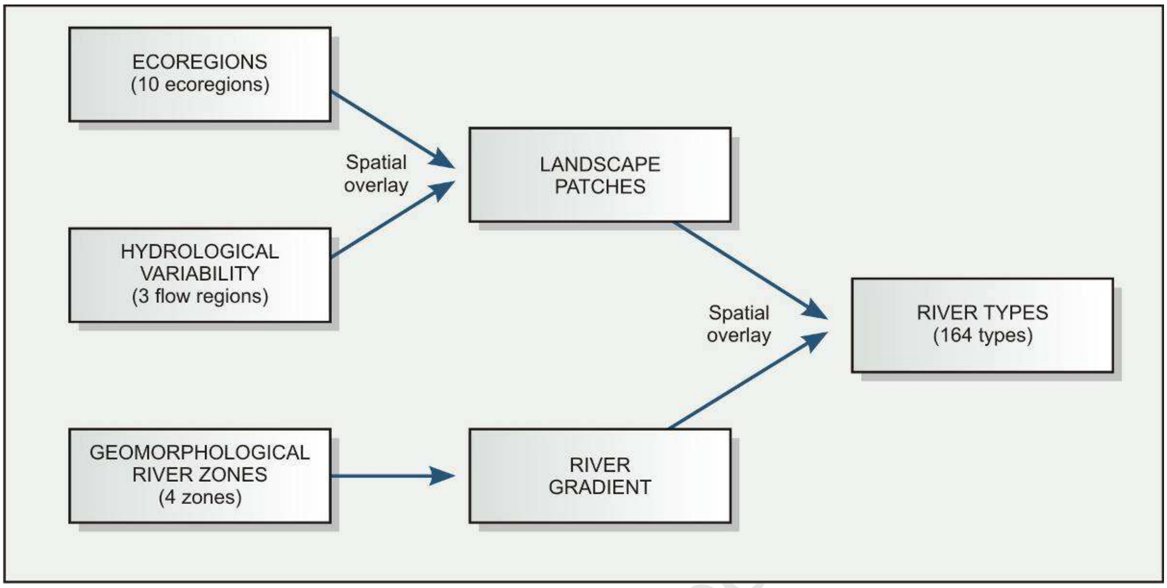

Figure 4.2. A schematic illustration modified from ***Roux et al. 2008****, showing the process of how river types were generated for this assessment. Three spatially explicit dataset sets on ecoregions, hydrological variability and geomorphology were spatial overlaid in a GIS to generate unique river types.*

4.3.2 Threatened fish species of the CFR

The diversity of fish species in the Cape Floristic Region is relatively low, but is characterised by a very high level (84%) of endemism (Impson & Cambray 2002). In this assessment, only species of special concern listed by the IUCN Red data list (2008) were considered. The species comprised: *Pseudobarbus afer*, *Pseudobarbus asper*, *Pseudobarbus burgi*, *Pseudobarbus tenuis*, *Pseudobarbus burchelli*, and *Barbus andrewi*. The data was obtained from the South African Institute of Aquatic Biodiversity (SAIAB). Even though all fish species occurring in the region were not included in the assessment, the persistence of all fish species in the study area was considered through the delineation of fish sanctuaries.

4.3.3 Wetland types

Wetlands are some of the least studied freshwater systems in the Cape Floristic Region; the data available is therefore relatively patchy. I relied on several datasets to derive a wetland database that spatially mapped all major and sensitive wetlands in the Cape Floristic Region. A broad wetland classification framework (Chapter 3) was adapted to capture the diversity of wetlands in the study area. To increase the level of confidence in the wetland classification and to generate suitable biodiversity surrogates, wetlands were classified based on ecoregions in an exercise similar to that of river types discussed above. As a surrogate for ecological processes special consideration was given to large wetland clusters of more than 1000 hectares, because large wetlands play a critical role in delivering ecosystems services such as flood and water quality regulation, and carbon sequestration (Gorham 2001). Targets for wetlands varied depending on wetland type, size and ecological integrity.

4.3.4 Estuaries

Priority estuaries in the CFR have been identified in a similar conservation planning exercise that focused on estuaries (Turpie & Clark 2007). In this analysis however, estuaries were not excluded because it's important that the priorities of the different freshwater components i.e. rivers, wetlands, and estuaries are aligned properly. The alignment of priority estuaries with rivers and wetlands is crucial for their persistence because rivers act as conduits for threats emanating from upstream. Since some estuaries had been included in the wetland database, those estuaries were considered as a wetland type and a target was set for each of them. The priority estuaries from Turpie and Clark (2007) were incorporated in the final stage of this assessment by highlighting those catchments they drain as priority catchments.

4.3.5 High water yield areas

Special consideration was given to high-water-yield areas in the Cape Floristic Region with the intention of ensuring that any potential priority conservation area should protect those high-water-yield areas critical for securing flow down the length of the river. The catchments that provide more than 50% of the mean annual runoff were assumed to be the catchments crucial for maintaining flow in the region. These catchments were flagged down, and a target of 50% was set to capture high water yield areas.

4.3.6 Fish sanctuaries

Fish sanctuaries are those areas in the region that are known to contain large fish populations occurring in relatively intact habitats. The sanctuaries were delineated based on expert knowledge (fish ecologists) and various fish databases available for the region. The fish sanctuaries were treated as a biodiversity feature in the assessment required to maintain fish species persistence. A minimum target of 20% set to capture these areas.

4.3.7 Fish migratory routes

Fish migratory routes were catered for by ensuring upstream-downstream connectivity between priority fish sanctuaries, based on expert knowledge. Specific species were not considered when migratory routes were being delineated, because fish sanctuaries are essentially regions with high fish populations. Establishing fish migratory routes based on broad criteria will also increase the potential for fish species persistence in cases of disturbances such as climate change, when even non-migratory fishes might become migratory to escape potential threats.

4.3.8 Ecological integrity of the rivers and wetlands

The ecological integrity of rivers is critical in systematic conservation planning, as a surrogate for biodiversity persistence. Rivers that are in good condition are more likely to support viable populations of species than rivers that have been transformed. Incorporating ecological integrity considerations in conservation planning is therefore a necessary measure required to minimise the vulnerability of the systems to anthropogenic change.

In this assessment the ecological integrity of river system was based on land use (Amis et al. 2007), and the present ecological condition class (PESC) (Kleynhans 2000). PESC is a measure of how the present ecological condition of a catchment has been modified from its natural reference state. The PESC assessment is based on aspects of water quality, flow, inundation cycle, stream bed condition, introduced instream biota, and the condition of the riparian zone. River systems in the study area were categorised into several classes depending on the ecological integrity. A river in a class A is one that is in intact condition with no alteration, while a river in class Z implies that the river system has been irretrievably degraded, and cannot be rehabilitated into an acceptable condition.

As with river systems, wetland ecological integrity was based on land use, and in this case a buffer of 200 metres was generated around each wetland and the percentage natural vegetation was determined within this 200m zone. The underlying assumption is that wetlands whose surrounding area is relatively intact are of high integrity and thus ecologically functional. The selection of priority areas was biased towards those systems with a relatively high ecological integrity to ensure biodiversity persistence.

4.4 The design of freshwater conservation area networks

4.4.1 Delineation of planning units (sub-catchments)

Sub-catchments were used as the planning units in this assessment, because of the importance of the catchment influence on in-stream integrity (Amis et al. 2007). Sub-catchments are also important for achieving lateral connectivity because they include riparian zones and all the areas draining a specific river reach (Moilanen et al. 2007, Nel et al. 2007). Sub-catchments were derived from digital elevation models (DEM) together with the 1:500, 000 river coverage of South Africa (DWAF 2004). The DEM was used to determine flow direction after the application of a series of algorithms to modify the DEM by imposing the linear river features onto them and filling in the sinks (Maidment & Morehouse 2002). The flow direction enabled the determination of flow accumulation and the delineation of sub-catchment boundaries around each river reach. In this assessment a river reach was defined as the stretch of river between confluences (Nel et al. 2006, Thieme et al. 2007). The fact that each sub-catchment represented a single river reach was very important for this assessment because it enabled the conservation planning software to deal with linear features, which is often a major challenge in conservation planning. The sub-catchment delineations gave rise to 864 units ranging in size from 184 to 4,134 ha.

4.4.2 MARXAN

Marxan was used in this analysis to identify a representative set of reserve networks that meet the set conservation goal for each biodiversity feature at a minimum cost to the area required (Pressey et al. 1993, Possingham et al. 2000). Marxan is a powerful tool for systematic conservation planning that is widely used for identifying priority areas for biodiversity conservation. It has mostly been used in marine and terrestrial ecosystems but is increasingly been adopted in freshwater ecosystems to identify priority areas (e.g. Linke et al. 2008, Amis et al. 2009). The major shortfall of Marxan for freshwater conservation planning is its inability to cater for the longitudinal nature of river systems, and the incorporation of upstream threats when identifying freshwater priority areas. The challenge of longitudinal connectivity was overcome by manually incorporating river systems that are upstream of a selected sub-catchment, and by using a Marxan functionality referred to as the “boundary length modifier” to ensure that the resulting set of reserve network is not disaggregated (See box for Marxan functionality). The freshwater priority areas were identified using an algorithm referred as simulated annealing in Marxan (Ball & Possingham 2000). Iterations for the model were set at 1billion, with a total of 20 runs, and a boundary length modifier of 0.2.

MARXAN FUNCTIONALITY

MARXAN uses an objective function to assign a total cost to a selected set of planning units depending on their relative importance as a potential reserve network. An optimisation method known as simulated annealing is then used to identify a near optimal set of planning units (i.e. potential conservation areas) which minimise the objective function (total cost). The objective function of MARXAN is described as:

$$\text{Total cost} = \sum_{PU} \text{PU costs} + \text{BLM} \sum_{PUs} \text{boundarylengths} \sum_{species} \text{CFPF} \times \text{Penalty} \times \text{Threshold penaly}$$

Where:

PU are the planning units (sub-catchments); Cost can be the area of the planning unit, the opportunity cost of selecting the planning unit, or an actual economic cost (e.g. the cost of land); The boundary length modifier (BLM) is a factor that ensures the selected planning units are spatially aggregated to enhance connectivity; the conservation feature penalty factor (CFPF) is a penalty for failing to represent a conservation feature; and the threshold penalty is associated with exceeding a set number of planning units or cost (Ball and Possingham, 2000).

4.5 Incorporating connectivity between freshwater priority areas

Freshwater ecosystems are linked laterally, longitudinally and vertically (Ward 1989), therefore effective conservation strategies must take into account how the different connectivity aspects of freshwater ecosystems are accounted for in conservation planning. Lateral connectivity refers to the association of lotic systems with their riparian zones, which result in the exchange of materials between them. In many cases stream organisms are also dependent on adjacent vegetation in the riparian zones (Palmer et al. 2005, Grimm et al., 2003). Longitudinal connectivity is related to the interconnectedness of river systems between upstream and downstream reaches. Longitudinal connectivity is important for processes such as migration and disturbances that emanate from upstream, and how they can be effectively managed to conserve biodiversity. The vertical aspect of connectivity in lotic systems is associated with groundwater, where some freshwater ecosystems are groundwater dependent (Sophocleous 2002). In this assessment vertical connectivity was not directly incorporated because there is no adequate information on groundwater recharge areas.

To incorporate both lateral and longitudinal connectivity into the conservation planning process, different measures were considered: the choice of planning unit, buffers, and manual incorporation of upstream catchments. Firstly by using sub-catchments as the planning unit, lateral connectivity was incorporated because when a specific biodiversity feature such as a wetland, a high water yield area or a river reach was selected, the entire catchment in which the specific biodiversity feature occurs was selected as well. Secondly, all priority river reaches were buffered by 200 metres on both sides, so as to secure their riparian areas. For longitudinal connectivity all river reaches upstream of a priority area were flagged as being critical for connectivity, and such river systems will be required to be maintained in a state that does not negatively impact on priority areas located downstream.

4.6 Expert input in the identification of priority freshwater ecosystems.

Freshwater expert opinion was sought extensively in the process of this assessment, right from the data collection stage to the review of the conservation planning outputs. Since the identification of the biodiversity priorities was mainly a desktop procedure, it was important that the outputs were thoroughly reviewed by experts who are familiar with the study area. A review workshop was convened to verify the results of the assessment, which led to the incorporation of a few areas into the final list of conservation priority areas that were not identified in the planning process, but were deemed to be very important by the experts. Most of the experts who assisted in the review were drawn from the fields of wetland ecology (wetland delineation and classification), fish biologists (identification of fish sanctuaries), invertebrate zoologists (identification of invertebrate hotspots), and landscape ecologists (landscape scale ecological processes).

4.7 Overlap between freshwater and terrestrial priorities

I used the two Water Management Areas (WMA) of the Berg and Breede to determine the extent to which terrestrial and freshwater priorities overlap. The Gouritz WMA was not included in the analysis because it was not covered by the terrestrial conservation plan used in the comparison. The conservation plan designed by Cowling et al. (2003) was adopted as the set of terrestrial priority areas. The terrestrial plan of Cowling et al. (2003) is of great practical significance because most conservation decisions in the CFR are based on it. It was therefore important to assess the extent to which freshwater priorities would be covered by that initiative. The Gouritz WMA was not included in the analysis to determine the overlap because part of that WMA falls outside the CFR and the terrestrial plan only covered the CFR.

4.8 Results

4.8.1 Conservation priority areas

The final list of proposed conservation priority areas required 148 of the total sub-catchments, excluding protected areas. In terms of area, 30.1% of the region should be provided with some form of protection if targets for the different biodiversity features are to be achieved. Of the areas to be set aside for conservation, 22.2% are in the priority catchments required to achieve all targets and the remainder (7.8%) are in catchments required to connect priority areas. Some of the rivers in the priority sub-catchments require restoration because they are in a very poor condition, and there were no options elsewhere to achieve targets. Rivers that require restoration comprised a total length of 661km, which is about 18% of the total length of priority rivers that were identified (Fig. 4.3).

4.8.2 Target achievement for biodiversity features

The physical classification of rivers in the CFR based on ecoregions, geomorphology and a hydrologic index gave rise to 164 river types. The ecological integrity of river systems varied widely (Fig. 4.4), with about 35% of the total length of rivers in class A or B. Rivers in class A or B are those with no significant anthropogenic disturbances and are deemed to be functional. Such rivers are surrounded by a minimum of 80% natural vegetation, which is critical for ensuring ecosystem function. Category C and D rivers (disturbed rivers, but with a rehabilitation potential), contributed 11% towards target achievement. River systems in categories EF to Z, with the worst anthropogenic disturbance accounted for 8% of the length river types required to achieve targets. Rivers in this category are degraded to such an extent that rehabilitating them may not be possible. The degraded rivers that were selected represent unique river types, if they were excluded from the analysis it would imply that targets for those river types would not be achieved.

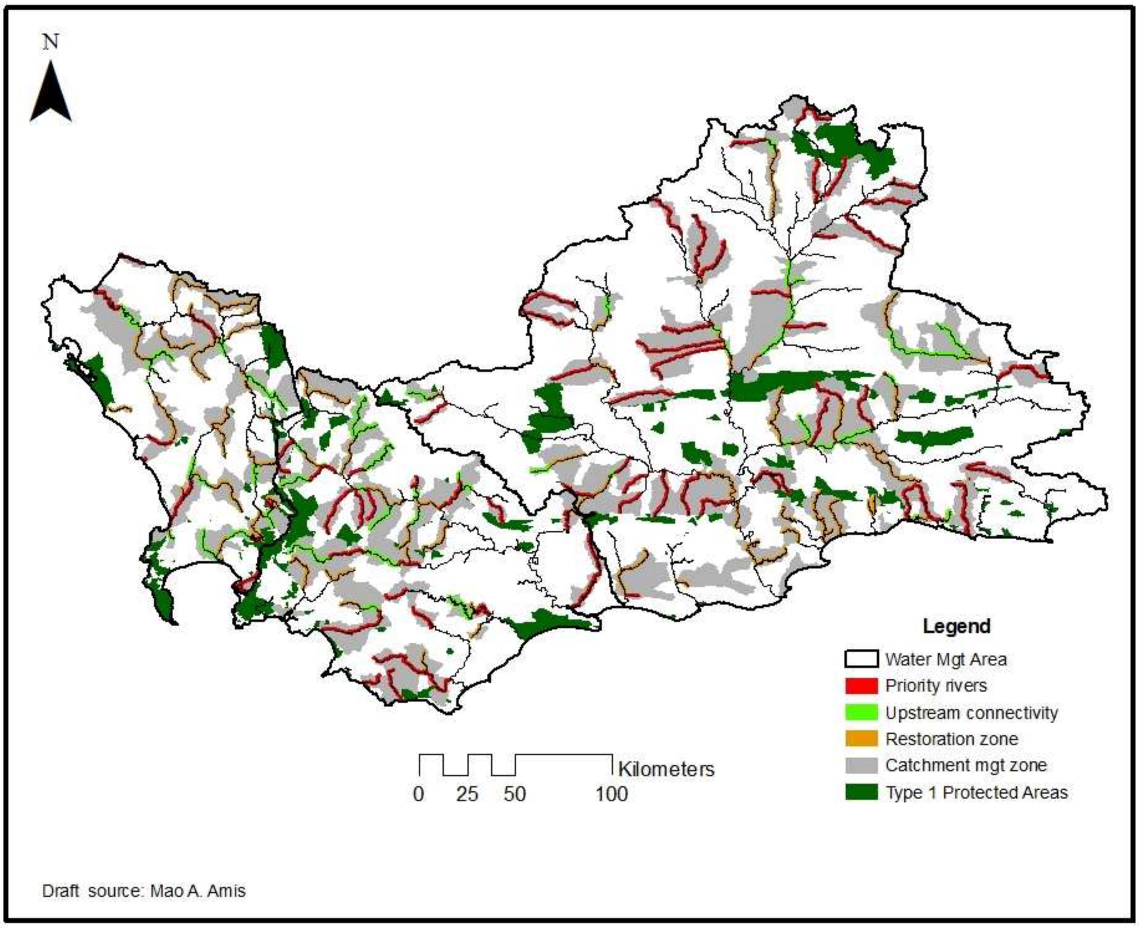

Figure 4.3. Map shows sub-catchments that were selected to achieve targets (best output). This is also the final portfolio that represents freshwater conservation priority areas in the CFR, including areas that will be required for connectivity. Freshwater focal areas are the rivers systems required to achieve targets; upstream connectivity represents additional river systems selected to achieve upstream connectivity, restoration zones represents unique river types that were selected as priority but required restoration, catchment management zones represent other biodiversity features such as wetlands, high water yield areas, and those catchments containing priority rivers.

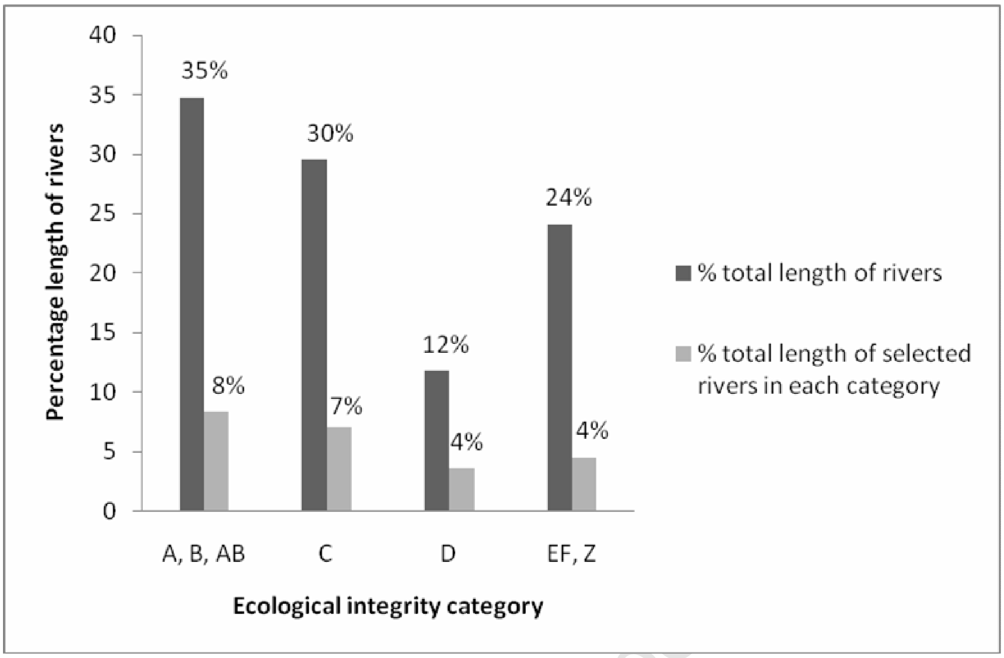

Figure 4.4. The ecological integrity of river systems showing the percentage total length of rivers in each ecological integrity category class, and the percentage length of rivers that there were selected as priority in each ecological integrity category.

The ecoregion level II classifications gave rise to 59 different wetland types, excluding those smaller than 0.2 ha in size. About 8.1% of the total area of wetlands in the region was included in the final list of potential conservation priority areas. The freshwater priority areas contained 7.9% of the total area of wetlands in the region, while the catchments required for connectivity contained 0.8% of the priority wetlands. In terms of the different wetland types, seepage and pan wetlands received the highest protection, with the catchment management zones comprising up to 14% of pans and 12% seepage wetlands. Lagoons were the least covered by the potential conservation areas with only 1% occurring in priority catchments. For estuaries and floodplains, 5% and 10% were covered by the priority areas respectively (Table 4.2).

Table 4.2. Freshwater targets achieved by the terrestrial priority areas in the Berg and Breed Water Management Areas.

Biodiversity features	% freshwater targets achieved by terrestrial plan
Fish species	15.3
Fish sanctuary	44.0
High water yield areas	43.7
River type	27.1
Wetland clusters	81.8
Wetlands	43.6

Due to data constraints on fish species in the study area, only fish listed as threatened in the IUCN Red data list (2008) were included. All targets were achieved for the six species that were included because targets were set at 100%. Where possible an effort was made to ensure that priority river reaches containing threatened fish species were connected to either downstream or upstream areas, which resulted in more areas to be set aside for conservation.

4.8.3 Target achievement for surrogates of ecological processes

Fish sanctuaries were delineated based on expert opinion to augment the six threatened species, by identifying sites where large fish populations are known to occur. Based on that assessment, 51 sub-catchments with a total river length of 1280km were selected as fish sanctuaries. All the demarcated fish sanctuaries could not be incorporated into the final list of conservation priority area, because it would require too much area to implement. Based on a target of 20%, the final list of conservation priority areas contained 26% of the fish sanctuaries that were identified by the experts. In addition all fish sanctuaries that were selected in the assessment were linked to other parts of the system through the establishment of migratory routes for known migratory fishes.

It was found that only 5% of the total area contributed 50% of the mean annual runoff. It was therefore critical that these high water yield catchments are provided with the highest protection possible. However, most of the high water yield areas also occured in mountainous region that are designated as mountain catchments, so they are relatively better protected than other biodiversity features in the study area. In addition 32% of the areas designated as high water yield areas were also selected as part of the conservation priority areas.

4.8.4 Incorporating connectivity and irreplaceability in the selection of priority areas

Longitudinal connectivity was achieved in this assessment by selecting all rivers upstream of priority catchments and flagging them as part of catchment management zones. Lateral connectivity was achieved through the inclusion of wetlands, especially floodplain and the valley bottom wetlands that are usually associated with rivers. Priority rivers were also buffered to create a riparian section that will require sustainable land-use practices in order to ensure that priority river reaches remain unperturbed.

Irreplaceability was measured as a score ranging from 1 to 100, with a score of 100 being the most irreplaceable. The selection of priority areas was biased towards sites with a high irreplaceability, as a result more than 56% of selected sub-catchments had an irreplaceability of between 80- 100 (Fig. 4.5). Irreplaceability is the likelihood that an area will be required to achieve targets (Ball & Possingham 2000, Linke et al. 2007). A good systematic plan should ensure that the conservation priority areas should include sites of high irreplaceability as far as possible.

4.8.5 Current protected area coverage of freshwater ecosystems

Results showed that 8.7% of the study area is already covered by type 1 protected areas (National and provincial parks, nature reserves and forest reserves) but the protected areas perform poorly in achieving targets for freshwater ecosystems (Fig. 4.6). Especially for fish species and sanctuaries, very low proportions of the targets were achieved by existing protected areas. This is a clear indication that when the protected areas were being set up, very little consideration was given to freshwater ecosystems.

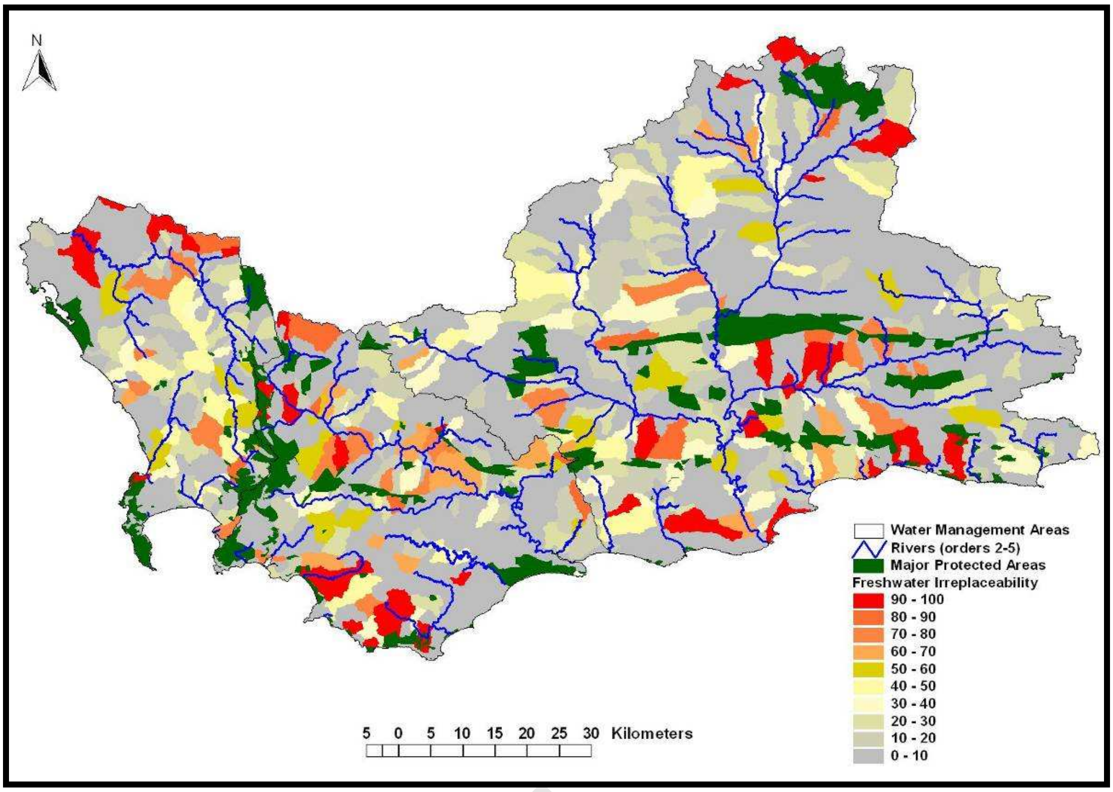

Figure 4.5. Irreplaceability map of freshwater sub-catchments in the Cape Floristic Region. A high irreplaceability score means that it is very likely such a subcatchment will be required to achieve targets.

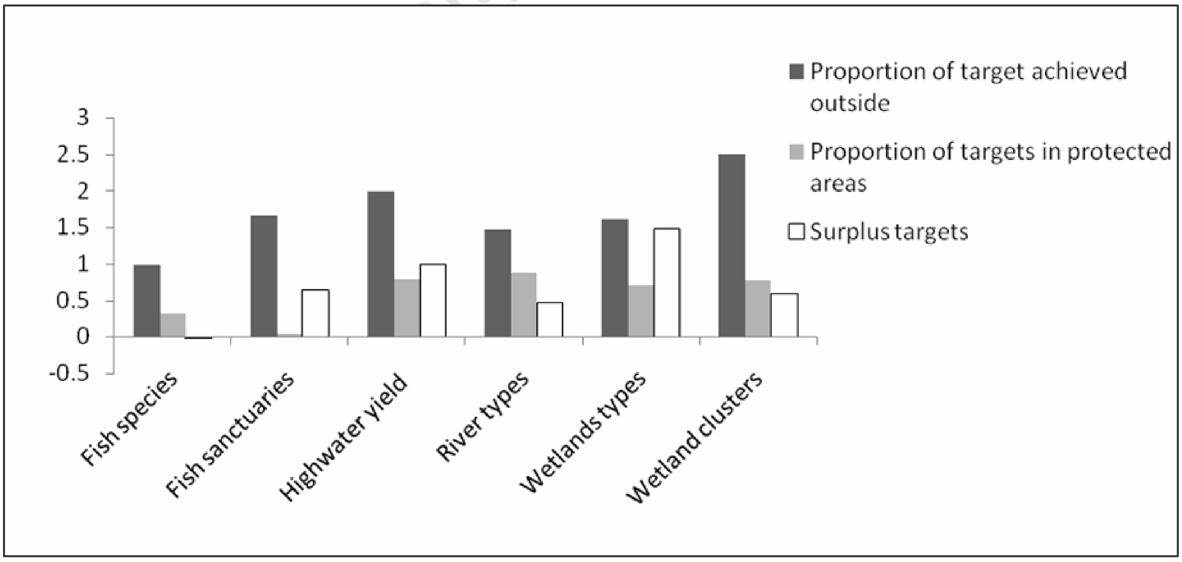

Figure 4.6. Target achievement for individual biodiversity features in inside and outside of type I protected areas in the Cape Floristic Region

4.8.6 Overlap between terrestrial and freshwater priority areas

The results from the analysis of the overlap between freshwater and terrestrial priority areas showed that there was a reasonable overlap of about 28% between the two systems (Fig. 4.7). This might be attributable to the fact that in their assessments Cowling et al. (2003) used rivers as corridors for linking terrestrial priority areas, and also included some fish species in their analysis. When target achievement for individual freshwater biodiversity features were assessed however, we found that targets for fish species and river types were very poorly covered by terrestrial priority areas (Table 4.3), probably because the terrestrial assessment did not consider river types, and did not give special consideration to threatened species.

Table 4.3. The protection levels of different wetland types in the proposed conservation priority areas, as represented in the selected priority catchments and the additional catchments required to maintain connectivity. The conservation priority areas represented about 8.1% of the total area of wetlands in the region.

Wetland type	% of wetland types in priority catchment	% of wetland types in catchment required for connectivity	% of total wetland type to be protected in the region
Estuary	5.1	0.1	5.2
Floodplain	9.1	0.9	10.0
Lagoon	0.9	0.00	0.9
Pan	12.2	0.1	12.3
Seep	13.2	1.2	14.4
Valley bottom	7.1	1.2	8.3
Other	8.5	1.8	10.3

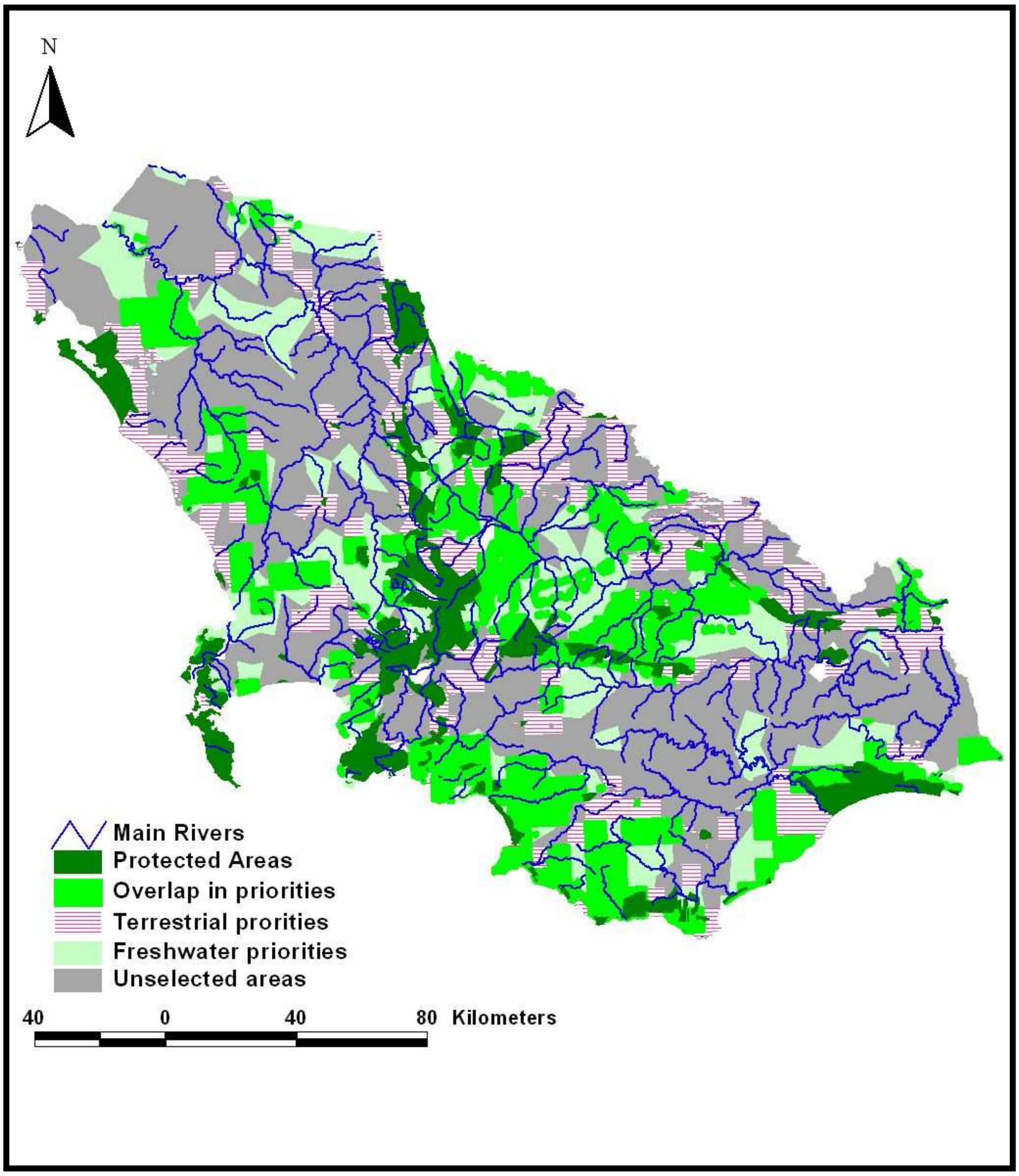

Figure 4.7. *The Berg and Breede Water Management Areas showing the focal freshwater priority areas, terrestrial priorities and the overlap between freshwater and terrestrial biodiversity priorities. The freshwater priorities shown above only represent the focal areas without incorporating additional areas that were flagged for connectivity purposes.*

4.9 Discussion

The Cape Floristic Region (CFR) has a long history of systematic conservation planning starting with Rebelo & Siegfried (1992), and since then novel approaches, such as the mapping of ecological processes (Rouget et al. 2003), and incorporation of climate change uncertainties into conservation planning (Williams et al. 2005) have been pioneered in the region. The major shortcoming of all initiatives in the CFR however, has been the failure to comprehensively incorporate freshwater biodiversity issues. Although this was the first comprehensive freshwater conservation plan for the region, it was very robust in identifying gaps prevalent in the protection of freshwater biodiversity in the CFR, and the additional areas that will be required to achieve freshwater conservation goals. The analysis of how much of the current network of terrestrial priority areas (Cowling et al. 2003) overlap with freshwater priorities gave an insight into the extent to which freshwater biodiversity issues are being addressed in the CFR. It was surprising to find that fish species and river types were very poorly covered by the terrestrial plan (Table 4.3), this is also consistent with Impson et al. (2002), that assessed the protection of indigenous fish by the protected area network in the CFR.

In terms of efficiency, about 30% of the area would be required to achieve all targets. Although the area required was a conservative estimate, it represents a high level of efficiency in the planning approach. Rivers straddle wide distances, and the need to maintain connectivity usually makes freshwater conservation plans to require extensive areas of land to be set aside. Due to competing land uses, large area requirements in freshwater conservation plans tend to make such plans impractical to implement. The challenge to implementing 'land-hungry' freshwater conservation plans could partly be addressed through the use of a hierarchical protection strategy (Abell et al. 2007), which was also used in this assessment. The zoning of the priority areas into different categories increases the potential for implementation, by ensuring that landuse guidelines reflect the specific requirement of each zone. For example the 'freshwater focal areas' will have the strictest landuse guidelines, while those catchments required for upstream connectivity will be less stringent. The delineation of restoration zones also highlighted areas in the landscape where urgent conservation intervention needs to be undertaken. Freshwater conservation plans that do not incorporate this kind of zoning could potentially create an implementation dilemma due to the impractical nature of freshwater conservation plans, and the lack of a clear protocol on where immediate conservation interventions should be undertaken.

A major challenge in freshwater conservation, which became apparent in this assessment, is the trade-off between biodiversity and resource utilization. Even though 35% of rivers in the region were in intact condition and categorised as class A or B, they only contributed of 8% towards target achievement. This implies that even though these rivers were in good condition, they are not diverse in terms of river types, and thus contributed minimally towards target achievement. But from a resource utilization perspective every freshwater system in good condition should be protected because of their importance in generating vital ecosystem services. This is a dilemma that has not yet been succinctly addressed in conservation planning and water resource management, with potentially negative implications for freshwater ecosystems. Most integrated water resource management frameworks are concerned with the importance of freshwater as a resource, while biodiversity conservation frameworks do not adequately address water use issues. An example in South Africa is the National Water Act (NWA), which has been deemed to be a global model of best practice in water legislation (Bohensky & Lynam 2005). Even though a provision for environmental flow requirements (ecological reserve) was specified in the NWA, it does not take biodiversity requirements into account. The ecological reserve requirement was designed to ensure that the freshwater systems continue to supply water without depleting the resource base (Grobler & Brown, personal comm.). The trade-off between freshwater biodiversity conservation and water as a resource is one that needs to be addressed if effective biodiversity conservation and sustainable water resource utilization are to be achieved.

Identifying freshwater priorities in the CFR was very challenging due to the limitations in available data. The variation in the scale at which data was collated, and the prevalence of data gaps in some regions, such as the upper part of the Gouritz WMA, required extensive data mining to standardize the datasets. In cases where datasets from different sources could not be reconciled the assessment relied on broad-scale biodiversity surrogates, or expert knowledge to fill the data gaps. A very useful by-product of this assessment was therefore the generation of extensive freshwater datasets for the region, which will serve as very useful templates for future conservation planning initiatives in the region. However, some of the datasets still require extensive field verification processes, such as the river types. Although the framework used to generate the river types is generally well accepted by experts in South Africa, field verification is still critical. The river typing was a desk-top exercise and is prone to accuracy related problems associated with GIS analysis such as poor alignments or false delineations. Because of the broad scale at which this assessment was carried out, data accuracy problems was not however, a major limiting factor. But if the same datasets were to be used for a fine-scale conservation planning, then more effort will be required to ensure that the level of confidence in the datasets used is increased.

It is a standard practice in systematic conservation planning to 'ground-truth' the set of identified priority areas (Cowling 2003). Due to resource constraints no ground-truthing was undertaken in this assessment. If the recommendations from this assessment are to be implemented then ground-truthing will be a pre-requisite. Further, in this assessment some rivers were listed as priorities even though they were of very low ecological integrity, because such rivers types are unique. Those rivers with low ecological integrity were flagged as those requiring restoration, but it has not yet been possible to ascertain whether they are restorable. The ground-truthing exercise should therefore focus on determining the restoration potential of such rivers. Those rivers that have been transformed beyond restoration should be removed from the list of priority areas, implying that these river types are effectively lost. The ground-truthing exercise should also focus on verifying the ecological integrity of priority wetlands. This assessment undertook a crude approach in determining wetland integrity based on the proportion of natural vegetation surrounding a wetland as a surrogate for wetland integrity. Assessing wetland integrity for conservation planning still remains a daunting task, but a field verification of the approach used to assess ecological integrity would be a useful endeavour as more advanced rapid assessment techniques for wetlands are developed.

CHAPTER 5. INTEGRATING FRESHWATER AND TERRESTRIAL PRIORITIES IN CONSERVATION PLANNING.

ABSTRACT

The integration of freshwater and terrestrial biodiversity priorities in systematic conservation planning is a major challenge to conservation planners. Maintaining upstream-downstream connectivity and the influence of catchments on freshwater ecological integrity are some of the issues that make it difficult to reconcile terrestrial and freshwater conservation planning. As a result most conservation assessments are often biased towards terrestrial systems without adequate incorporation of freshwater biodiversity in determining priority areas for conservation. In this paper, a protocol is proposed for integrating the assessment of freshwater and terrestrial priorities in conservation planning, based on a case study from Mpumalanga Province in South Africa. The approach involves the separate assessment of freshwater priority areas, and using the outcome to influence the selection of terrestrial priority areas. This allowed both freshwater and terrestrial biodiversity to be incorporated in conservation planning without compromising their unique requirements. To test the effectiveness of this approach, we assessed percentage overlap between freshwater and terrestrial priority areas, target achievement, and the area required to achieve targets. I then compared the outcome from the proposed approach with the separate assessments of freshwater and terrestrial biodiversity priorities, and when both systems are given an equal weighting in a single assessment. The results showed that there was a noticeable improvement in the overlap of priority areas for freshwater and terrestrial biodiversity from 23% to 47%. Target achievement for freshwater biodiversity improved by 10% when terrestrial assessment was based on freshwater priority areas as opposed to terrestrial systems being assessed alone. There was negligible increase in area required, whether there was integration of freshwater and terrestrial biodiversity or no integration. It is concluded that the most efficient way to achieve integration in conservation planning is to preferentially select areas where freshwater and terrestrial biodiversity priorities overlap.

***Keywords**:* *Freshwater and terrestrial biodiversity, conservation assessment, integration, overlap*

5.1 Introduction

Freshwater biodiversity are under threat from anthropogenic disturbances and global change, and a large number of species are threatened with extinction *(*Lake et al. 2000, Saunders et al. 2002*)*. Systematic conservation planning can play an important role in reversing the threats to biodiversity, by identifying biodiversity priorities and developing conservation strategies in an efficient and defensible manner (Margules & Pressey 2000, Sarkar et al. 2006, Pressey & Bottrill 2008). Current conservation planning procedures however, are not comprehensive in identifying priority areas across multiple systems (Abell 2002). Typically, terrestrial, freshwater and marine biodiversity priorities are assessed separately, undermining their interdependence (e.g. Fairbanks et al. 2001, Fox & Beckley 2005, Strange et al. 2007). Although good examples are very rare, joint biodiversity assessments result in an efficient identification of priority areas for conservation (e.g. Kremen et al. 1999, Noss et al. 2002). Such joint assessments enable linkages between freshwater and terrestrial systems to be maintained, and duplication of effort is minimised. However, integrated approaches to conservation planning must recognise the differences between freshwater and terrestrial systems, and take into account the need to keep their assessment separate to some extent but achieve integration in those aspects where there is an overlap.

Key differences that render freshwater systems unique include the longitudinal nature of river systems and the associated connectivity (Dudgeon et al. 2006), which make rivers act as conduits for threats emanating from upstream. Rivers are also constrained by catchments, which imply that they are affected by both local and upstream conditions as water flows from one catchment to another (Dudgeon et al. 2006, Durance et al. 2006). These unique characteristics of freshwater systems result in considerable complexity when planning for freshwater and terrestrial systems together (Abell 2002, Sowa et al. 2005, Suski & Cooke 2007). It is partly for these reasons that integrated approaches to conservation planning need to recognise the differences between freshwater and terrestrial systems when identifying joint biodiversity priorities, to effectively deal with these challenges.

Freshwater conservation planning driven by decision support systems (DSS) typically involves the collation of data in the planning domain that represents the best available information on biodiversity in the region. Biodiversity is mapped at a specific scale at which priorities will be identified referred to as a planning unit. Depending on the objectives of the planning exercise and the scale at which data is available, the planning unit could range from the scale of a river reach to a catchment. Once the data has been collated this information is fed into a conservation planning software under specific criteria, and based on repeated iterations the planning tool will identify a set of places in the landscape that best achieve the set criteria. When the focal areas have been identified then issues of connectivity can be incorporated at a later stage to link up the focal areas to other parts of the landscape. The use of DSS in freshwater conservation planning is therefore limited, compared to terrestrial conservation planning where biodiversity priorities can be fully identified based on DSS tools alone. The challenges to integrated conservation planning are therefore twofold, based on the differences in the ecology of freshwater and terrestrial systems, and on the technical aspects of conservation planning.

The lack of freshwater specific tools could potentially be overcome by innovatively applying approaches that have been developed for terrestrial ecosystems. Studies have shown that tools for terrestrial conservation could lend themselves to application in freshwater ecosystems (Dunn et al. 2003, Nel et al. 2008). Many of the terrestrial tools however, require modification to suit the unique challenges presented by freshwater ecosystems. For example Moilenen et al. (2007), modified the basic functionality of the of the conservation planning tool Zonation (Moilenen et al. 2006), to account for a directional upstream-downstream connectivity in freshwater ecosystems. Similarly Linke et al. (2008) adapted a complementarity based algorithm to address upstream connectivity issues in river systems, which enabled the selection of upstream catchments in addition to local assemblages when identifying priority areas for conserving freshwater biodiversity. The above examples show that current conservation planning tools could potentially address the challenges of integrating freshwater and terrestrial priorities in conservation planning, if applied innovatively.

Even though there are no there are no specific approaches that have been developed for the integration of freshwater and terrestrial priorities in conservation planning, many examples of how integration has been achieved exist. Broadly, integration can be categorised into four types: incidental, partial, complete and additive integration. *Incidental integration* may be achieved when a conservation plan is designed for only one system (e.g. freshwater or terrestrial), but biodiversity targets for the other systems are incidentally achieved due to an overlap in their priority areas. For example Maitland and Lyle (1992), found that in Great Britain, National Nature Reserves offer protection to more than 30 fish species, even though the nature reserves were not designed with the intention of protecting freshwater biodiversity. In a reversed example from Portugal Filipe et al. (2004) identified priority areas for fish conservation at the scale of catchments. Even though terrestrial biodiversity were not considered in the assessment, it is very likely that a catchment selected for fish conservation could be important for terrestrial biodiversity as well. Freshwater hierarchical frameworks (Higgins et al. 2005, Abell et al. 2008) that emphasis both fine and coarse scale filters to conservation planning also have a very high potential of achieving integration, because the coarse filter approach highlights landscape scale processes that may encompass terrestrial biodiversity, while the fine filter approach addresses freshwater specific needs. In general, incidental integration is dependent on the chance occurrence of an overlap between freshwater and terrestrial priority areas, it is therefore not very appropriate for achieving integration.

A terrestrial conservation plan that includes some elements of freshwater biodiversity or vice versa in its assessment can be referred to as *partial integration*. There are many examples of partial integration such as the incorporation of wetlands in terrestrial assessments (e.g. Lombard et al. 1997), the selection of a littoral forest based on its association with other habitats such as mangroves, marsh and permanently flooded forests (Kremen et al. 1999), and preferentially selecting areas adjacent to protected areas (Thieme et al. 2007). *Complete integration* is a result of assessing priorities for all the different systems simultaneously (e.g. Noss et al. 2002). In this approach the selection of priority sites is not biased towards a specific system as the biodiversity features of both freshwater and terrestrial systems are given equal weighting. Finally, the most widely used approach is *additive integration*, where freshwater and terrestrial biodiversity are assessed separately and the resulting output is combined as the set of priority areas (e.g. Sowa et al. 2005).

This paper proposes a step-wise approach for achieving integrated freshwater and terrestrial biodiversity priorities in conservation planning. The main emphasis is on the need to recognise the inherent differences between freshwater and terrestrial systems, which is achieved by asking the following questions:-

i) Is the separate assessment of freshwater and terrestrial systems an efficient way of identifying biodiversity priorities?

ii) How can freshwater and terrestrial priorities be effectively integrated in systematic conservation planning?

5.2 Methods

5.2.1 The proposed approach for integration

I adopted a two-step protocol that recognised freshwater and terrestrial systems as operating differently, and was able to identify integrated biodiversity priorities for both systems without compromising their individuality. The two-step protocol consisted of: -

i) a separate assessment of freshwater systems followed by
ii) assessment of terrestrial systems being driven by freshwater priority areas

This approach enabled both freshwater and terrestrial priorities to be successfully integrated, while at the same time recognising the uniqueness of each system in the assessment. I used the systematic conservation planning software MARXAN v1.8 (Ball & Possingham 2000) with the user interface CLUZ (Smith 2004) to carry out the analysis, although it could equally be applied using other systematic conservation planning tools.

5.2.2 MARXAN as a tool for achieving integration

MARXAN is a widely used tool in systematic conservation planning to identify priority areas for biodiversity conservation (Sarkar et al. 2006). It offers several algorithms that can be used to solve problems of minimum representation in conservation planning, to generate a solution set that optimally achieves the set objectives (Ball & Possingham, 2000). In this study the simulated annealing algorithm was used to identify a near optimal set of planning units that achieves goals for both freshwater and terrestrial biodiversity simultaneously. The MARXAN functionality can be described as:

$$\text{Total cost} = \sum_{PU} \text{PU costs} + \text{BLM} \sum_{PUs} \text{boundarylengths} \sum_{species} \text{CFPF} \times \text{Penalty} \times \text{Threshold penaly}$$

Where:

PU are the planning units (grid cells or sub-catchments); *cost* can be the area of the planning unit, the opportunity cost of selecting the planning unit, or an actual economic cost (e.g. the cost of land); The boundary length modifier (*BLM*) is a factor that ensures the selected planning units are spatially aggregated to enhance connectivity; the conservation feature penalty factor (*CFPF*) is a penalty for failing to represent a conservation feature; and the *threshold penalty* is associated with exceeding a set number of planning units or cost (Ball & Possingham 2000).

The planning unit cost feature of MARXAN influences the selection of planning units. Planning units with low cost are selected preferentially to those with a higher cost if options for achieving targets exist. In most conservation assessments using MARXAN, the cost function is used to minimise the area required to achieve biodiversity targets. In such cases the area of the planning unit is used as a cost, and the goal is to represent a specified number of biodiversity features at minimum cost (i.e. area) (e.g. Oetting et al. 2006, Shriner et al. 2006, Carwardine 2007). The cost function could also be specified in monetary terms, such as the cost of purchasing the land if it were to be set for a reservation or the anticipated earnings from fisheries that would be forgone if a marine network were to be established. Stewart and Possingham (2005) derived the commercial value of each planning unit based on the total rock lobster catch (kg) for each individual planning unit. They then used a combination of the commercial value and the planning unit area to derive a cost input for MARXAN, with the goal of minimising the total rock lobster catch displaced by the marine reserve system and reserve system area.

I used the cost function of MARXAN in a novel way to integrate freshwater and terrestrial biodiversity priorities, by using the resultant irreplaceability from the freshwater assessment as the cost input for the terrestrial assessment (Fig. 5.1). Irreplaceability is simply the likelihood that a specific planning unit will be required to achieve the set conservation targets (Ball & Possingham 2000, Linke et al. 2007). The higher the irreplaceability value, the more likely that such a planning unit will be required to achieve targets.

The cost input for the terrestrial assessment was thus given as:

$$X = (100-A) + B$$

Where: -

X= Terrestrial cost input; (100-A) = Standardized freshwater irreplaceability (i.e. the higher the irreplaceability, the lower the cost); B= Habitat transformation.

Based on this formula, planning units with a high freshwater irreplaceability were given a lower cost in the terrestrial assessment. MARXAN was then able to bias the selection of planning units for the terrestrial assessment, towards those planning units that were untransformed and of high freshwater irreplaceability, if options existed. In this way freshwater concerns were partially integrated when terrestrial biodiversity priorities were being assessed.

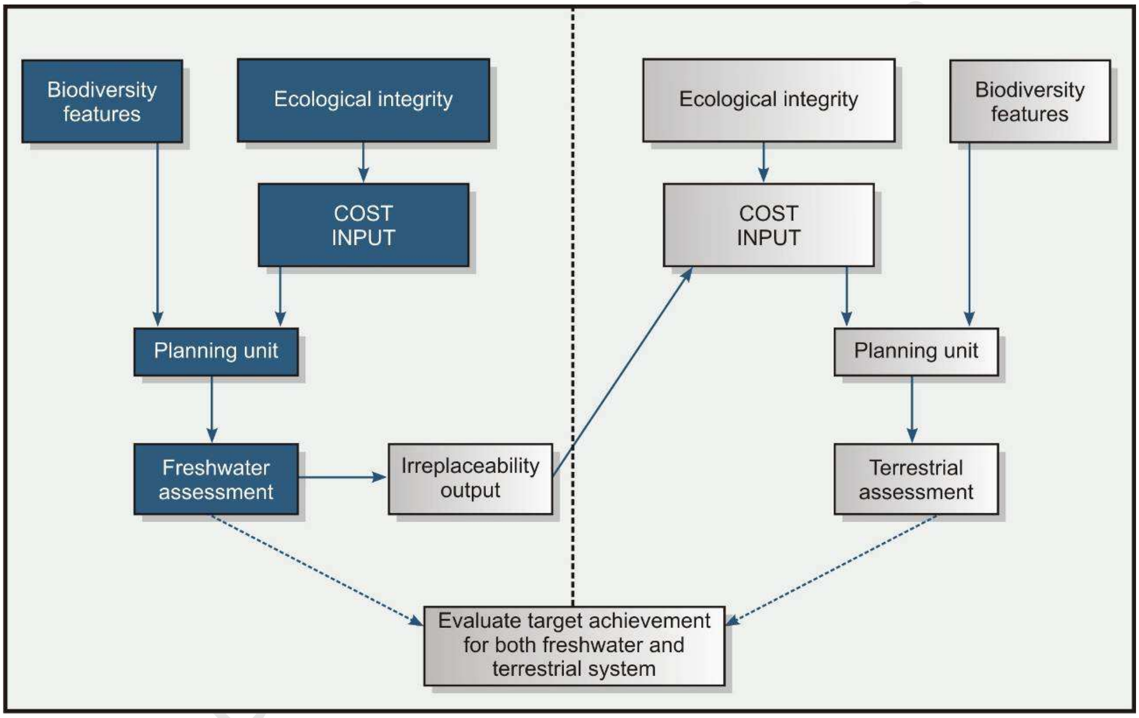

Figure 5.1. A schematic illustration of how integration of freshwater and terrestrial biodiversity could be achieved. Dark blue boxes show the initial step that involve the separate assessment of freshwater biodiversity priorities, and how those priorities are used to drive the assessment of terrestrial biodiversity priorities (grey boxes).

5.2.3 Study Area: Mpumalanga Province, South Africa

The Mpumalanga province located in the northeast of South Africa (Fig. 5.2) is 87,000 km^2 in extent (6.5% of South Africa's area). Mpumalanga is an important region for conserving both freshwater and terrestrial biodiversity in southern Africa, as it forms part of the source of four major river systems and comprises three of the nine terrestrial biomes found in South Africa. In terms of freshwater biodiversity, the Crocodile River system alone contains 49 fish species (RHP 1998) and is one of the most productive catchments in South Africa in terms of water provision (DWAF 1995). Mpumalanga is also drained by other major river systems which include the Oliphants, Orange, Inkomati, and Pongola river systems (DACE 1999). In terms of terrestrial biodiversity, grasslands are some of the most threatened systems in South Africa (Reyers et al. 2007).

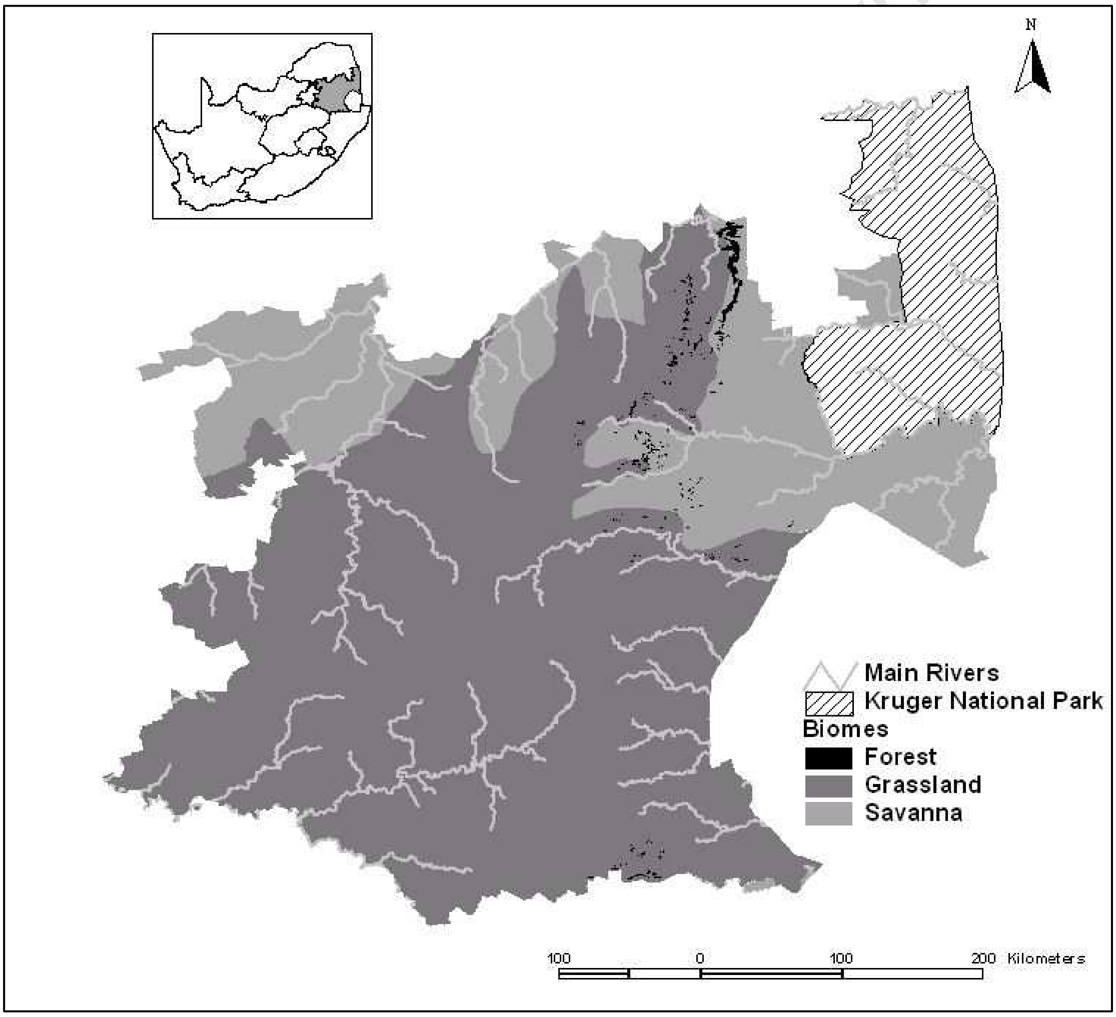

Figure 5.2. Location of the study area showing major terrestrial biomes, rivers and the Kruger National Park, inset is the map of South Africa.

Despite the fact that 24% of Mpumalanga is under some form of protection, many critical biodiversity areas are still inadequately represented in the protected area network (Roux et al. 2008, Ferrar & Lotter, 2007). The biggest threats to biodiversity in the region are habitat loss, invasive alien species and water demand exceeding water availability. The region has also been experiencing a drastic decline in water quality in the last 6 years (DACE 1999). Mpumalanga has a population of 3.1 million people and its main economic activities are agriculture and mining. It is also home to the famous Kruger National Park, and tourism is a significant contributor to the economy of the province.

5.2.4 The Design protocol

A biodiversity feature is defined as either the biotic or abiotic component of the system that determines its structure, function and composition (Noss 1990). In this study a total of 157 freshwater biodiversity features (Table 5.1) and 340 terrestrial biodiversity features were mapped (Table 5.2). Biodiversity targets for both freshwater and terrestrial biodiversity were set through an expert workshop, where specific criteria were used to formulate quantitative targets for each biodiversity feature. For example fish species had a target range of between 50%- 100% of fish populations depending on the conservation status of the fish species based on the IUCN red data list. For rivers a flat target of 20% of the total length was set for each river type (Roux et al. 2008), consistent with other freshwater conservation planning exercises (e.g. Nel et al. 2007, Thieme et al. 2007). Terrestrial vegetation targets varied, depending on the inherent species diversity within each vegetation type (Table 5.2).

Hexagons and sub-catchments were used as planning units for the terrestrial and freshwater assessments respectively. The area was subdivided into 64,000 hexagons with a size of 118 hectares, and 1503 sub-catchments with an average size of 2500 hectares. Sub-catchments were delineated from a digital elevation model (DEM), and the 1:500, 000 river coverage for South Africa (DWAF 2004). The conservation planning tool (MARXAN) was set to 100 runs, with 1 billion iterations and a boundary length modifier of 0.2 for the terrestrial assessment. The cost input to MARXAN varied according to the specific approach (Table 5.3). The threshold penalty factor was set to 0, and all assessments were performed using simulated annealing with a two-step iterative improvement (Ball & Possingham 2000).

Table 5.1. Biodiversity features used to identify freshwater priority areas.

Biodiversity Feature	Description	Extent/Size	Target
Rivers – biodiversity	Rivers classified into Ecoregion Level 2 types as surrogate for biodiversity	28 666 km of rivers, 29 types, stream orders >2	20% flat target in line with the national spatial biodiversity assessments (NSBA)
Seepage and valley bottom wetlands	Wetlands classified into functional types, then according to ecoregion level 2 biodiversity template	Only wetlands 0.2 ha or greater; 113 628 individual wetlands; totaling 312 771 ha. representing 51 types (functional and biodiversity)	A baseline target of 15% and adjusted upwards according to biodiversity value
Pan wetlands	Scale of 1: 50 000. Classified into perennial and non-perennial pans, and ecoregion level 2 biodiversity template	23922 ha of pans; 39 pan types	Base target of 50%, adjusted upwards according to biodiversity value
Peat wetlands	Point records indicating the presence of wetlands	77 point records	Base target of 80% for known peat wetlands
Fish species	Known distribution of 4 threatened fish species	4 fish modeling known distribution; 908 km of river length	Target based on rarity and distribution, ranged from 50- 100%
Important pan clusters	Buffer all pans with 1 km and remove transformed areas. Include clusters 500 ha or greater, with area to perimeter ratio > 300	132 611 ha.	Base target of 50% for all important wetland and pan clusters
Important wetland clusters	Buffer all wetlands greater than 5 ha with 1 km and remove transformed areas. Include clusters 1 000 ha or greater, with area to perimeter ratio > 300	266 820 ha.	Base target of 50% for all important wetland and pan clusters
High water yield areas	Median annual runoff per quaternary catchment. Select catchments producing 50% of Mpumalanga runoff as high water yield areas	High water yield areas equate to 15% of the planning domain	Base target of 50%

Table 5.2. Summary of biodiversity features used to identify terrestrial priority areas

Biodiversity Features	**Description**	**Exten/size**	**Target**
Vegetation/forest types	68 vegetation types: National vegetation types other than forests (biodiversity surrogates	68 types: 9 forest; 28 grassland; 31 savanna	19%- 28% for vegetation, 59.5-71.7% for forests
Amphibians	Modelled distribution of important species	3 species	1.2- 84.5%
Birds	16 threatened species (known, modelled and/or nesting sites- 24 features in total	Feeding and know sites- 19 species, Nesting sits- 7 species	7- 100%
Invertebrates	Buffered known localities and point localities	17 species	75%
Mammals	Modelled distributions, actual distributions and buffered sites	13 species	0.5- 100%
Plants	Known point localities	187 species	100%
Reptiles	Modelled distributions	10 species	1.2- 84.5%
Special features	Selected pans and wetlands with unique biodiversity; all natural caves	Point records identify wetland and pans with unique features. Caves have 250m buffer	100%
Surrogates for ecological processes	Key landscape features maintain ecological and evolutionary centred on biological movement and connectivity	Escarpment/summit corridors; centres of endemism; montane and Highveld grassland patches, forest patches	1.17- 100%

Table 5.3. Alternative approaches for assessing freshwater and terrestrial priorities in conservation assessments, showing the cost input and planning units used for the analysis.

Approach	Description	Cost input	Planning unit
1) Freshwater alone	***Incidental integration.*** Freshwater priorities were assessed independently, then we determined how much terrestrial targets are incidentally achieved	PESC* + habitat transformation within sub-catchment	Sub-catchment
2) Terrestrial alone	***Incidental integration.*** Only terrestrial priorities are determined and then the target achievement for freshwater under terrestrial assessment was evaluated.	Habitat transformation	Hexagons
3) Freshwater driven by terrestrial priorities	***Partial integration.*** Terrestrial priorities were used as a constraint to influence the selection of freshwater priorities. It enabled areas of overlap to be preferentially selected in the freshwater assessment	Terrestrial irreplaceability + habitat transformation	Hexagons
4) Terrestrial driven by freshwater priorities	***Partial integration.*** Freshwater priorities were used as a constraint to influence the selection of terrestrial priorities. It enabled areas of overlap to be preferentially selected in the terrestrial assessment	Freshwater irreplaceability + habitat transformation	Hexagons
5) Freshwater and terrestrial assessed together	***Complete integration.*** Freshwater and terrestrial biodiversity were given equal weighting in a single assessment.	PESC+ habitat transformation	Sub-catchments
6) Freshwater and terrestrial added after separate assessments	***Additive integration.*** Freshwater and terrestrial priorities were assessed independently and the separate outputs added up as the set of priority areas	N/A	

*Present ecological status condition

5.2.5 Evaluation of the proposed approach for integration

Results from the proposed approach were compared with various alternative approaches (see Table 5.3). The comparison was undertaken to determine whether the use of freshwater priorities areas to inform terrestrial conservation assessment (partial integration) was a better approach for achieving integration compared to the other approaches such as separately assessing freshwater and terrestrial priorities (additive integration). To evaluate the effectiveness of the proposed approach for integration, we assessed the area required to achieve biodiversity targets, target achievement, the degree of spatial overlap between outcomes of alternative approaches (Table 5.3) and the correlations between freshwater and terrestrial irreplaceability scores. The integration of freshwater and terrestrial priorities was regarded as effective, if biodiversity targets were met with minimum area requirement. All the different approaches for integration (Table 5.3) were evaluated using the criteria discussed above. We then compared the outcome from the approach with the partial approach for integration that we have proposed (using freshwater priorities to inform the identification of terrestrial priorities).

I did pair-wise comparisons (degree of spatial overlap) between alternative approaches to evaluate the effectiveness in the alignment of the spatial priority areas (the areas required to achieve all targets) for both freshwater and terrestrial biodiversity. The best output from MARXAN was then used to compare between the alternative approaches. The Coefficient of Similarity/overlap was used in the same way as (van Jaarsveld et al. 1998, Warman et al. 2004), and was calculated as:

$$O_s = n_T / (n_T + n_j + n_i)$$

Where: -

n_T = area of overlap between freshwater and terrestrial systems
n_j = area selected for the freshwater system only
n_i = area selected for the terrestrial system only

5.3 Results

5.3.1 Area required to achieve biodiversity targets

No major differences were found in area required to achieve biodiversity targets with or without integration of freshwater and terrestrial conservation priorities. When terrestrial systems were assessed based on our suggested approach of partial integration (terrestrial driven by freshwater priorities), 36.6% of the area was required, compared to 35.6% when terrestrial priorities were determined without any consideration of freshwater biodiversity (incidental integration). The area required also increased marginally to 38.3%, when targets for both freshwater and terrestrial systems were met in a single assessment (complete integration) (Fig. 5.3).

5.3.2 Target achievement

Target achievement for freshwater biodiversity improved from 48%, when terrestrial systems were assessed independently (incidental integration), to more than 57% using our proposed approach (terrestrial driven by freshwater priorities) (Fig. 5.3). The approach could not achieve freshwater targets by 100%, because this was essentially a terrestrial assessment and freshwater systems were merely used to influence the process of identifying terrestrial priority areas.

5.3.3 Overlap between freshwater and terrestrial priority areas

Priority areas for both freshwater and terrestrial biodiversity were generally better aligned using our proposed approach (terrestrial driven by freshwater priorities), compared to the separate assessment of each system (incidental integration) or when both were given equal weighting in a single assessment (complete integration). When freshwater and terrestrial biodiversity were assessed independently, there was a 23% overlap in their priority areas. Using the proposed approach, the area of overlap between freshwater and terrestrial priority areas improved noticeably from 23% to 47% (Table 5.4). When terrestrial priorities were used to influence the selection of freshwater priorities, however, it did not result in a noticeable alignment of priority areas for both systems (Fig. 5.4).

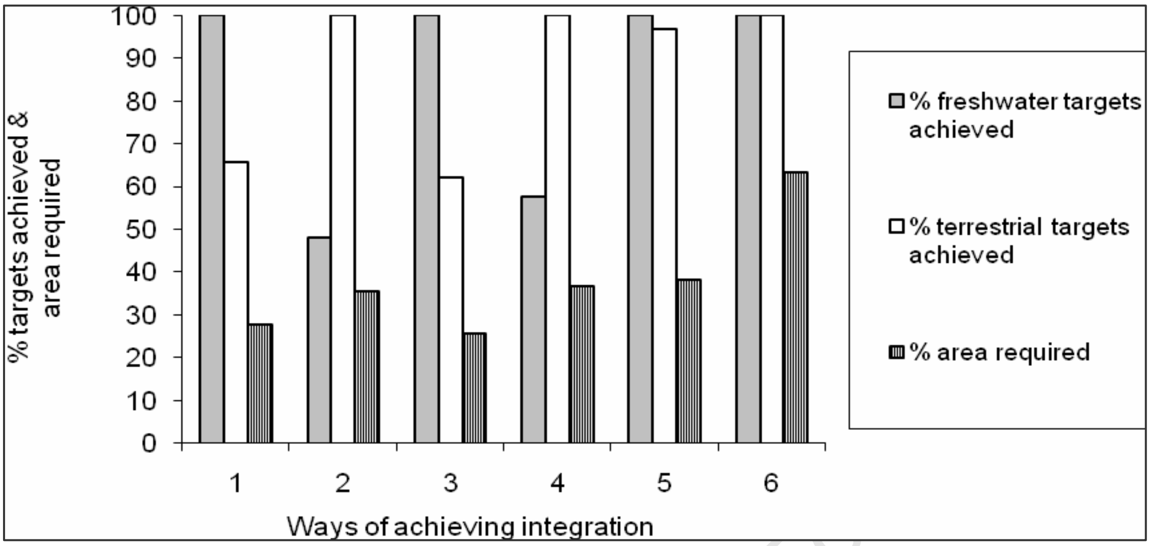

Figure 5.3. Target achievement for terrestrial and freshwater biodiversity under the six different approaches tested. (1= Freshwater alone, 2= Terrestrial alone, 3= Freshwater driven by terrestrial, 4= Terrestrial driven by freshwater, 5= Freshwater and Terrestrial together, 6= combined output of freshwater alone and terrestrial alone). The bar graphs show the extent to which each of the approaches was able to achieve targets for both freshwater and terrestrial biodiversity..

Table 5.4. Spatial overlap and correlations in areas required to achieve targets and irrelaceability scores respectively. The analysis was performed on the different approaches for integration, where Jaccard's coefficient of similarity was used to calculate spatial overlap. Pearson correlation was performed between irreplaceability scores, all were correlations were significant at <0.05.

	Overlap between priority areas		**Correlation between irreplaceability**	
Approach	***Freshwater alone***	***Terrestrial alone***	***Freshwater alone***	***Terrestrial alone***
1) Freshwater alone	-	23%	-	0.1
2) Terrestrial alone	23%	-	0.10	-
3) Terrestrial driven by freshwater	47%	-	0.31	-
4) Freshwater driven by terrestrial priorities	-	25%	-	0.31
5) Freshwater and terrestrial	45%	37%	0.45	0.42
6) Freshwater + terrestrial (additive)	-	-	-	-

Figure 5.4. *Solution sets (priority areas) generated by the different approaches used to assess freshwater and terrestrial biodiversity.* ***1) = Freshwater alone; 2) = Terrestrial alone; 3)= Freshwater driven by terrestrial; 4)= Terrestrial driven by freshwater; 5)= Freshwater and terrestrial assessed together.*** *The maps represent the 'Best Output' from MARXAN, and it represents the optimum solution that achieves all the targets for the system being assessed. For example map 2 shows the areas that are required to achieve all targets for terrestrial biodiversity.*

5.3.4 Correlation between terrestrial and freshwater irreplaceability

A visual inspection of the irreplaceability outputs from the different approaches (Fig. 5.5) clearly showed a low correlation between freshwater and terrestrial outputs, when both were assessed independently (incidental integration). This was confirmed when correlation analysis were performed between the different approaches (Table 5.4). When freshwater priorities were used to influence the terrestrial assessment, correlation between the two systems improved significantly.

5.4 Discussion

I have described a method to identify integrated biodiversity priorities for both freshwater and terrestrial systems, using a decision support system (MARXAN) and demonstrated its application in a region of South Africa. Although the entire approach is based on a specific MARXAN functionality, the same principles could be applied in other conservation planning tools such as Zonation (Moilanen & Kujala 2006). The main strength of this approach is the ability to identify focal areas where freshwater and terrestrial biodiversity priorities overlap, and yet keep the assessment of both systems separate to cater for their unique requirements. The approach for integrating freshwater priorities in conservation planning varies from other approaches because it is automated. Automating this process is very important as systematic conservation planning is getting widely embraced as the preferred approach for identifying biodiversity priority areas compared to other traditional approaches such as scoring. Previous attempts to achieve integration of freshwater and terrestrial biodiversity priorities did not use systematic conservation planning tools. For example Weitzell et al. (2003) separately identified freshwater and terrestrial biodiversity priorities, and manually selected 50 sites where freshwater and terrestrial priorities overlap. In a remote Amazonian basin Thieme et al. (2007), preferentially selected freshwater priorities areas that were adjacent to protected areas, using an approach that was largely manual. By automating the process of integrating freshwater and terrestrial priorities using this proposed approach, required the independent collation of data for both systems, which allowed me to generate multiple optimum solutions using various criteria (Table 5.3). This flexibility is important in systematic conservation planning, because it allows different scenarios and the priority areas to be evaluated as more data becomes available (Abellan et al. 2005).

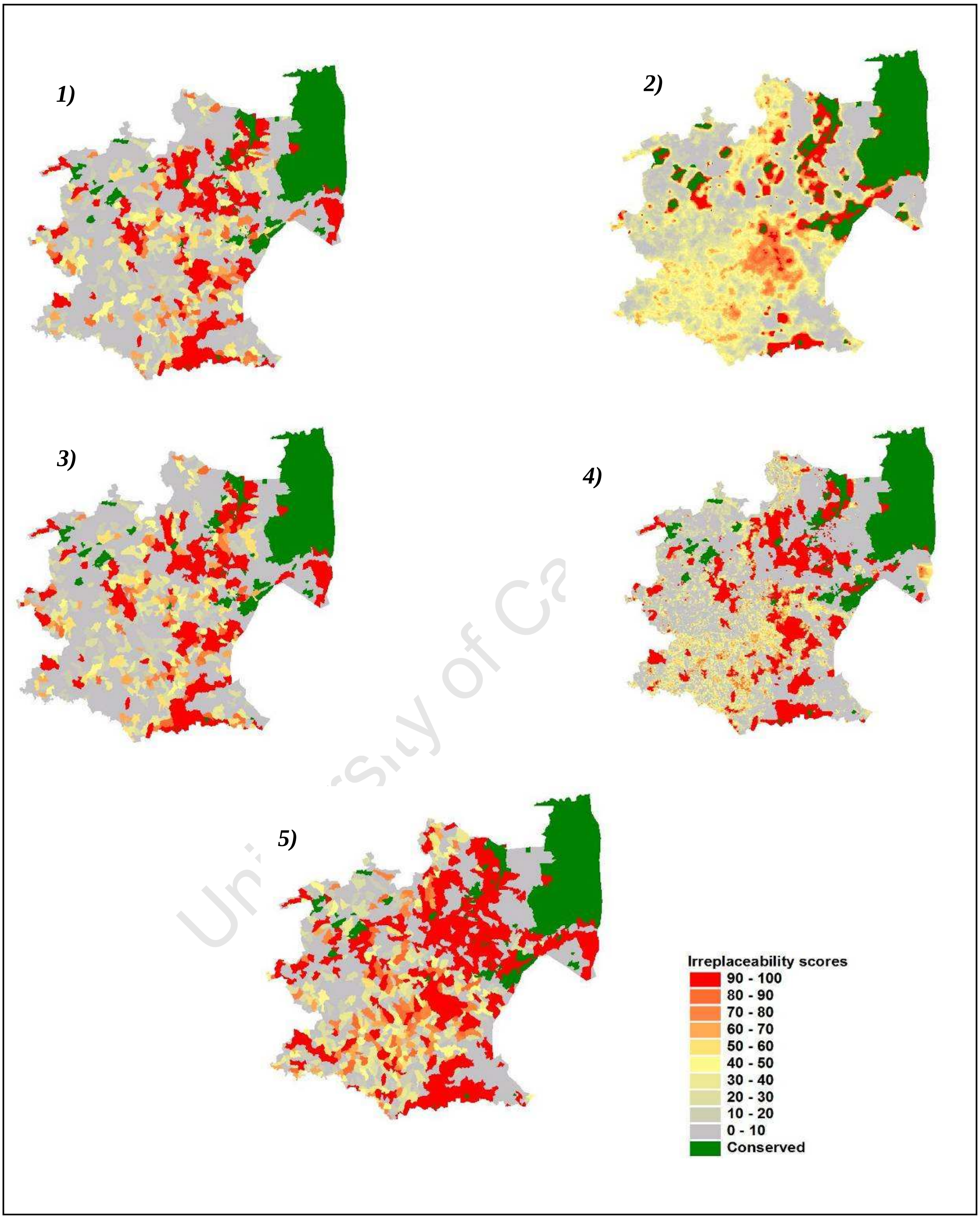

Figure 5.5. *Irreplaceability maps from the different approaches used to assess freshwater and terrestrial biodiversity.* ***1) = Freshwater alone; 2) = Terrestrial alone; 3)= Freshwater driven by terrestrial; 4)= Terrestrial driven by freshwater; 5)= Freshwater and terrestrial assessed together****. Areas with a high irrepleaceability values are those with a high likelihood of being required to achieve targets*

The overall goal of this assessment was to identify a suit of conservation areas that optimally achieve the targets for biodiversity pattern and process (Margules & Pressey 2000). As is the case in many regions globally, data on freshwater biodiversity is often patchy (Revenga & Kura 2003). In this assessment there were only 157 freshwater biodiversity features compared to 340 terrestrial biodiversity features. The important thing however, was that the best available data on freshwater biodiversity was collated in this assessment, which was supplemented in some cases with abiotic surrogates (Thieme et al. 2007, Roux et al. 2008), and modelled data. Wetlands were also comprehensively mapped and incorporated as freshwater biodiversity features, with special consideration of large wetland clusters that support vital landscape scale ecological processes (Gorham 2001). A minimum target of 15% for freshwater biodiversity features ensured that most of the targets were achieved when freshwater biodiversity priorities were assessed. The 4 threatened fish species listed in the IUCN Red data list (2004) had a target of 100%, which gave rise to a total length of 908 km of rivers that were selected. The terrestrial biodiversity assessment relied on a comprehensive database that comprised of more than 60 vegetation types, and numerous species. Issues of biodiversity persistence for both freshwater and terrestrial ecosystems were incorporated using a number of measures, such as restricting target achievement for freshwater biodiversity to healthy catchments (Nel et al. 2008), and including special features like centres of endemism, escarpment/summit corridors, and forest and grassland patches as surrogates for terrestrial biodiversity processes (Cowling et al. 2003).

The approach adopted in this study takes cognisance of the likelihood that for a long time, the conservation agenda will still be driven by terrestrial biodiversity. This is because freshwater systems are perceived to be of ‘less importance’ to warrant the setting up of freshwater conservation areas, and research in this sector is still viewed to be less attractive (Abell 2002). The approach therefore seeks to influence the process of terrestrial biodiversity assessments so as to be more efficient in capturing freshwater biodiversity concerns as much as possible. It is therefore acknowledged that giving equal weighting to freshwater and terrestrial biodiversity would have been a more effective approach for achieving both freshwater and terrestrial conservation targets (Fig. 5.3). Joint assessment of both freshwater and terrestrial biodiversity would however, undermine the uniqueness of each system, and make it impossible to incorporate at a later stage, vital ecological processes required by each system for biodiversity to persist. Unlike in terrestrial systems where conservation plans can be fully designed using DSS tools, freshwater conservation planning is still at its infancy and DSS tools are still wholly inadequate for designing freshwater conservation areas (Thieme et al. 2007). Therefore any attempt to identify integrated freshwater and terrestrial priorities should allow the incorporation of freshwater features not dealt with in a DSS environment. Another important consideration is that freshwater and terrestrial biodiversity often have different stakeholders both in terms of management and resource users, hence the importance of keeping the assessment of freshwater and terrestrial biodiversity separate at some stage in the process. Our approach is therefore a compromise between a scenario where freshwater and terrestrial biodiversity are assessed independently of each other (resulting in low efficiency), or lumping both freshwater and terrestrial biodiversity together (undermining their uniqueness).

The proposed approach does not deal comprehensively with some major challenges in freshwater conservation planning such as connectivity, and threats emanating from upstream catchments. This approach however, achieved lateral connectivity through the use of catchments as the planning unit (Linke et al. 2008), and it incorporated a surrogate measure for ecological integrity based on land use (Amis et al. 2007), and the present ecological status condition (PESC), of each catchment. The inability to comprehensively address connectivity and upstream threats was a limitation mainly imposed by the functionality of current conservation planning tools, although recently some attempts have been made to address the same (see Linke et al. 2008, Moilanen et al. 2008). Because of this limitation, other critical freshwater ecological processes are often incorporated at a later stage after the critical focal areas (Abell 2007), have been identified, hence the importance of keeping freshwater and terrestrial assessments separate. In the design phase where issues of freshwater connectivity are incorporated based mostly on expert knowledge present an additional opportunity to 'bump up' target achievement for freshwater biodiversity.

Due to competing land uses, the size of an area designated as priority for conservation purposes plays a critical role in successful implementation of conservation initiatives. One of the important aspects of systematic conservation planning is the attempt to minimize the size of the potential conservation area. Integrating freshwater biodiversity into this process was initially thought of as adding "several layers of complexity to an already complicated effort"

(Abell 2002, Pg. 1437), both in terms of the planning process and the area that may be required. In this paper, we were able to show that by incorporating freshwater biodiversity in a simple way, both freshwater and terrestrial conservation goals could be adequately achieved and the area required minimised, without complicating the process of systematic conservation planning.

The analysis performed in this paper is one of the few that use DSS tools to integrate freshwater and terrestrial biodiversity assessment, but is by no means exhaustive. Issues that might still need to be addressed include scale mismatches and the application of the approach in an entirely different region to test its universal applicability. Terrestrial biodiversity tends to be mapped at a finer scale than freshwater biodiversity and in certain situations this might present a problem. I however, found the use of simple GIS rules to downscale solution sets of freshwater priorities identified at the catchment scale to the scale of the terrestrial assessment very effective. This analysis was based in a region of equal importance for both freshwater and terrestrial biodiversity, but the dynamics of integration might play out somewhat differently in a region that is of critical importance to only one system. In cases where freshwater and terrestrial biodiversity issues are both critical like in our case study, this approach presents a potentially useful way to handle integration of freshwater and terrestrial biodiversity assessment.

CHAPTER 6. ASSESSING MANAGEMENT EFFECTIVENESS IN IMPLEMENTING FRESHWATER BIODIVERSITY PRIORITIES IN SOUTH AFRICA.

***Question**: Are there enabling mechanisms for the effective implementation of freshwater biodiversity priorities in South Africa?*

ABSTRACT

Systematic biodiversity planning has become a critical process in developing effective strategies for biodiversity conservation, because it enables optimum use of limited conservation resources. However, the practice of systematic biodiversity planning is marred by an implementation crisis, where in many cases biodiversity plans fail to be translated into concrete on ground conservation action. In this study, management effectiveness of key implementing agencies with a mandate to protect freshwater biodiversity was undertaken to understand their effectiveness in implementing freshwater biodiversity. The Study was undertaken in Crocodile and Marico Water Management Area (WMA), in the North West/Gauteng Provinces of South Africa. Current effective mechanisms for implementing freshwater biodiversity priorities include good regulatory mechanisms, shared biodiversity values, good learning basis and the existence of adequate monitoring and communication. Barriers to effective implementation include misaligned biodiversity conservation strategies, inadequate capacity, and inadequate alignment of monitoring and data. In order to develop effective strategies for implementing freshwater biodiversity priorities there is a need to incorporate management effectiveness measures into conservation planning frameworks to understand the barriers to implementation.

***Keywords**: Management effectiveness, freshwater ecosystems, implementation, biodiversity priorities, conservation planning*

6.1 Introduction

Conservation Biology is often referred to as a crisis discipline (Given 1993, Pullin & Knight 2001, DeSalle & Amato 2004), due to the increasing threats to biodiversity and species extinction. Coupled with the 'biodiversity crisis' are the limited resources for carrying out conservation action (Balmford et al. 2002, Saterson et al. 2004, Salzer & Salafsky 2006). This predicament gave rise to concepts such as ecosystem management (Grumbine 1994) and systematic conservation planning (Margules & Pressey 2000), as it became apparent that the limited resources must be prioritised in an efficient manner to maximise conservation effort (Myers et al. 2000).

To develop effective conservation strategies, systematic conservation planning should be viewed as a continuum from assessment to implementation. A conservation plan cannot be adequate without an effective implementation strategy (Knight et al. 2006), because the ultimate goal of systematic conservation planning is to protect biodiversity based on a process of objective and defensible decision-making (Margules & Pressey 2000). In the light of the limited conservation resources, the need to devise effective implementation strategies is therefore a key goal of conservation planning.

Most conservation planning studies are still focused on refining biodiversity assessment techniques however, with less emphasis on developing robust implementation strategies (Knight et al. 2006). The bias of systematic conservation planning towards biodiversity assessments, more especially in terrestrial and marine ecosystems has led to an 'implementation crisis' (Knight & Cowling 2003). Even South Africa, which is regarded as a leader in the field of systematic conservation planning (Balmford 2003) is still battling with how to effectively implement their biodiversity plans. Many conservation plans ranging from broad to fine scale (e.g. Cowling et al. 2003, Ferrar & Lotter 2007) have been developed in the country, but the challenge that still remains is how to translate those plans into tools for making sound conservation decisions.

The challenges of implementing conservation plans can partly be attributed to inadequate forward thinking on how implementation would proceed after a biodiversity assessment has been completed. For example a lack of understanding of the strengths and weaknesses of the existing mechanisms for implementing biodiversity priorities could prove a major hurdle in translating conservation planning products into useful biodiversity conservation tools. This situation is more pertinent in freshwater ecosystems, where the mandate for protecting freshwater ecosystems is multi-sectoral (Mackay & Ashton 2004, Pahl-Wostl et al. 2007). In freshwater conservation planning, stakeholder buy-in and the existence of a culture of cooperation between agencies responsible for protecting freshwater biodiversity is therefore critical when planning for freshwater ecosystems (Mackay & Ashton 2004, Roux et al. 2008). Due to the complexity of freshwater ecosystems and divergent stakeholder expectations, there are no easy solutions to the implementation of effective conservation strategies. It is therefore important to view freshwater ecosystems as socio-ecological systems where social learning by all actors takes place (Pahl-Wostl 2002). When freshwater conservation is undertaken as a learning process, the spirit of collaboration and cooperation can be entrenched among key actors in the sector (Roux et al. 2008).

In order to develop effective implementation strategies there is a need to understand the key drivers to the implementation crisis, through the documentation of case studies (Knight et al. 2006). Assessing the management effectiveness of organisations mandated with the protection of freshwater biodiversity offers a useful insight into how implementation strategies should be developed based on the strengths and weaknesses of current mechanisms. The evaluation of management effectiveness refers to how well conservation objectives are being reached (Hocking et al. 2000). In relation to protected areas, where it has mostly been applied (e.g. Hockings et al. 2003, Goodman 2003, Ervin 2003), the evaluation of management effectiveness addresses three key areas: 1) the suitability and design of the protected area or network, 2) appropriateness of management systems and, 3) the achievement of conservation objectives of the protected area (Hocking et al. 2000). To date measures of management effectiveness have seldom been applied outside protected area settings, yet they could be very useful for assessing the extent to which conservation objectives have been attained.

In the case of freshwater ecosystems it is necessary to adopt the practice of assessing management effectiveness outside of protected area setting because decisions in water resource management involve complex political processes (Pahl-Wostl et al. 2007). Adopting robust approaches for evaluating management effectiveness of freshwater ecosystem is therefore critical for achieving conservation objectives. The application of the concept of management effectiveness has however rarely been applied in freshwater ecosystems with a few exceptions such as Ramsar sites (Pavese & Burgess 2008). The limited adoption of management effectiveness techniques for evaluating freshwater ecosystems could be attributable to their complexity, which makes it difficult to decide on what to measure, where, and how. A useful way of measuring the achievement of freshwater conservation objectives would be to assess the effectiveness of key institutions mandated with the protection of freshwater biodiversity. The institutional evaluation would give an insight into the current strengths and weaknesses in the mechanisms for conserving freshwater biodiversity.

In South Africa the National Water Act (NWA) 1998 ushered in opportunities for new and innovative approaches to water resource management, and has been lauded in many cases as one of the most progressive water legislations globally (Bohensky & Lynam 2005). The NWA makes water for basic human needs a human right and also considers environmental sustainability in managing water resources (Wynberg 2002). The NWA therefore offered an opportunity for new ways of water resource management, including equity in water distribution and mainstreaming of biodiversity in the water resource management agenda. Because the NWA was a precedent-setting legislation however, its implementation became a major challenge because there were no concrete examples and experiences to show how the legislation could be implemented (Rogers et al. 2000). As a result, in the decade since the NWA came into effect the water sector in South Africa has been going through a learning curve.

The NWA led to devolution of water resource management from central government to local authorities (Rogers et al. 2000, Pollard & du Toit 2007). Under this arrangement the country was divided into 19 Water Management Areas (WMAs), each managed by a Catchment Management Agency (CMA), the principle authority through which water resource management is to be administered. The establishment of the CMAs offers a good opportunity for freshwater biodiversity issues to be incorporated into catchment management strategies. This is important because for a long time water resource management was primarily concerned with supply rather than with an integrated approach to water management (Pollard & du Toit 2007). In this study the potential for freshwater biodiversity priorities to inform catchment management strategies was investigated, by assessing the management effectiveness of current institutions mandated with the conservation of freshwater biodiversity in an important Water Management Area in South Africa.

6.2 Method

6.2.1 Study area and the Organisations that were assessed

South Africa is divided into 19 Water Management Areas (WMA) for ease of managing freshwater resources. This study was undertaken in the Crocodile (West) and Marico WMA, hereafter referred to as Crocodile and Marico WMA. The WMA forms the hub of the economic activities in South Africa, where the cities of Johannesburg and Pretoria are located. Water demand for both domestic and industrial use is therefore considerable. Agriculture, mining and light industries are the main activities in the WMA (DWAF 2002). Freshwater biodiversity is facing enormous pressure from anthropogenic disturbances in the WMA and mining in particular has negative influence on the ecological integrity of freshwater ecosystems in the WMA (Amis et al. 2007).

The evaluation of management effectiveness focused on the main implementing agencies in the WMA with a mandate of protecting freshwater resources. A multi Agency assessment approach was adopted because freshwater ecosystem management falls across multiple jurisdictions. No Catchment Management Agency (CMA) has yet been established in the WMA, as a result only regional government departments and nature conservation agencies are responsible for protecting freshwater resources, and they comprise:-

- Department of Water Affairs & Forestry (DWAF) Regional Offices, Gauteng Province
- Department of Water Affairs & Forestry (DWAF) Regional Offices, North West Province
- Gauteng Province Department Of Agriculture, Conservation & Environment (GDACE)
- North West Province Department Of Agriculture, Conservation & Environment (NWGDACE)
- North West Parks And Tourism (NWPARKS)
- South African National Biodiversity Institute (SANBI)

6.2.2 The approach used

The assessment was carried out as part of a broader project that sought to develop a motivational/reflective assessment tool for inter-agency cooperation (Roux et al. 2009). The assessment tool was developed in conjunction with various experts in ecology and social science with considerable input from conservation practitioners, to validate its content. The tool is based on a scorecard format similar to various frameworks that have been developed for assessing management effectiveness in protected areas such as the Parks in Peril scorecard (TNC 2000), the WWF tracking tool (Ervin 2003b), and the World Commission on Protected Areas Framework (Hockings et al. 2000).

The assessment tool was developed to promote inter-agency cooperation through social learning; it was therefore designed for self evaluation between partner organisations. The scorecard tool is still undergoing refinement, with the aim of applying it in other WMAs in the Country. This study was based on results from a preliminary assessment to test the suitability of the tool.

The questionnaire comprised of 30 questions divided broadly into 5 main parts (see appendix 1 for detailed questionnaire):-

- Context (where are we now?)
- Planning (Where do we want to be?)
- Monitoring (What data are we collecting and how?)
- Management (how do we want to go about making a difference?), and
- Co-learning as a cross-cutting aspect

All organisations were assessed in a single interactive assessment workshop, which drew representatives from each organisation directly mandated with the protection of freshwater resources in the WMA. It was hoped that the interactive workshop environment would foster co-learning among partner organisations through self criticism and feedback. In this study the six organisations assessed were represented by a total of 9 individuals, who are in charge of freshwater protection in their respective organisations.

During the workshop the facilitator announced each question, and the participants chose a score appropriate to their perception. The questionnaire was arranged on a likert-type scale ranging from 0 (very poor) to 3 (excellent). Participants were encouraged to reflect before choosing a particular score, and in some cases they were asked to justify their scores, making the qualitative assessment as objective as possible. The assessment did not focus on outputs and outcomes, because it was assumed that suitable management systems would result in the effective implementation of biodiversity priorities, although other externalities beyond the jurisdiction of implementing agencies might also influence biodiversity outcomes.

A spreadsheet facility was used to automatically compute responses so that the outcome of the evaluation could be shared among participants during the workshop to allow participants reflect on their performance. Results of the assessment were computed using descriptive statistics. For ease of interpretation, scores were categorised as follows: 1) Score $\leq$ 0.5 (Low), 2) 0.5 < Score $\leq$ 1.5 (Fair), 3) 1.5 < Score $\leq$ 2.5 (Good), 4) 2.5 < Score (High). The degree to which the different organisations perceive the same issue provides insight into how common the issue is to the organisations. A high "commonality" (or consensus) suggests they all perceive the issue in the same way. A low commonality suggests they perceive the issue very differently. This is useful because a strategy to address a problematic issue may be different in the two circumstances. Accordingly, the degree of commonality was also determined by simply calculating the difference between the highest and lowest score, to form the following categories: *1) Range = 0 (High commonality), 2) Range = 1 (Good commonality), 3) Range = 2 (Fair commonality), 4) Range = 3(Low commonality).*

6.3 **Results**

Overall the implementing agencies evaluated appear to be performing well (Fig. 6.1), but a lot could be done to improve their performances further. The key issues that emerged from the evaluation gave an important insight into how freshwater ecosystems can be effectively managed. Some issues were unique to each organisation, while other concerns were common to organisations.

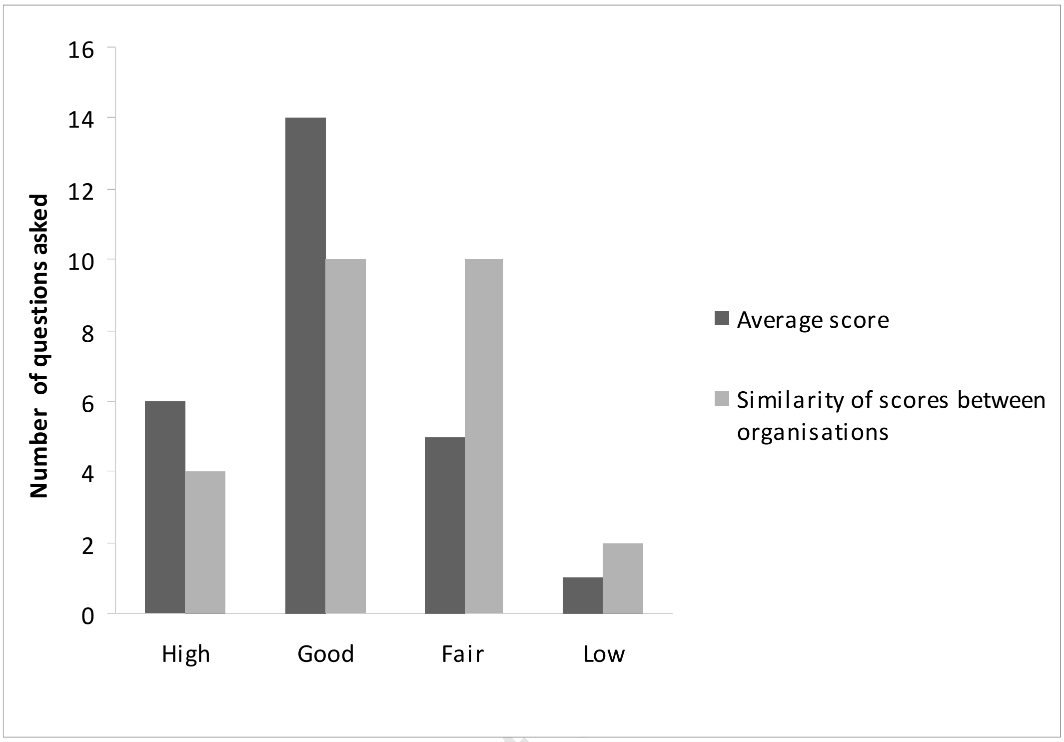

Figure 6.1. The average score of the response of organisations to the questions posed, and the similarity in scoring between the different organisations. On average most organisations evaluated were 'good' in their performance, while the similarity in responding to the same questions varied between 'good' and 'fair'.

6.3.1 Strengths of the current mechanisms in implementing freshwater biodiversity plans

6.3.1.1. Good regulatory framework

Participants reported the existence of adequate regulatory mechanisms and the good understanding of their own mandates (score 2), aided in some instances by the existences of relevant forums such as the wetland forum. It was perceived that sufficient legislative and legal mechanisms were in place for the conservation of freshwater biodiversity (Fig. 6.2). The appropriateness of existing statutes was rated highly by all the organisations (score=3); there seems to be a good understanding of the current legislation governing freshwater ecosystems in South Africa. In relation to systematic conservation planning, it was found that decision-making was in many instances informed by the conservation plan in the region. Most notable was the use of conservation plans as tools for making decisions on development proposals.

Because conservation plans highlight important biodiversity areas in the landscape, they were regarded as useful tools for making decisions on proposals for changing land use.

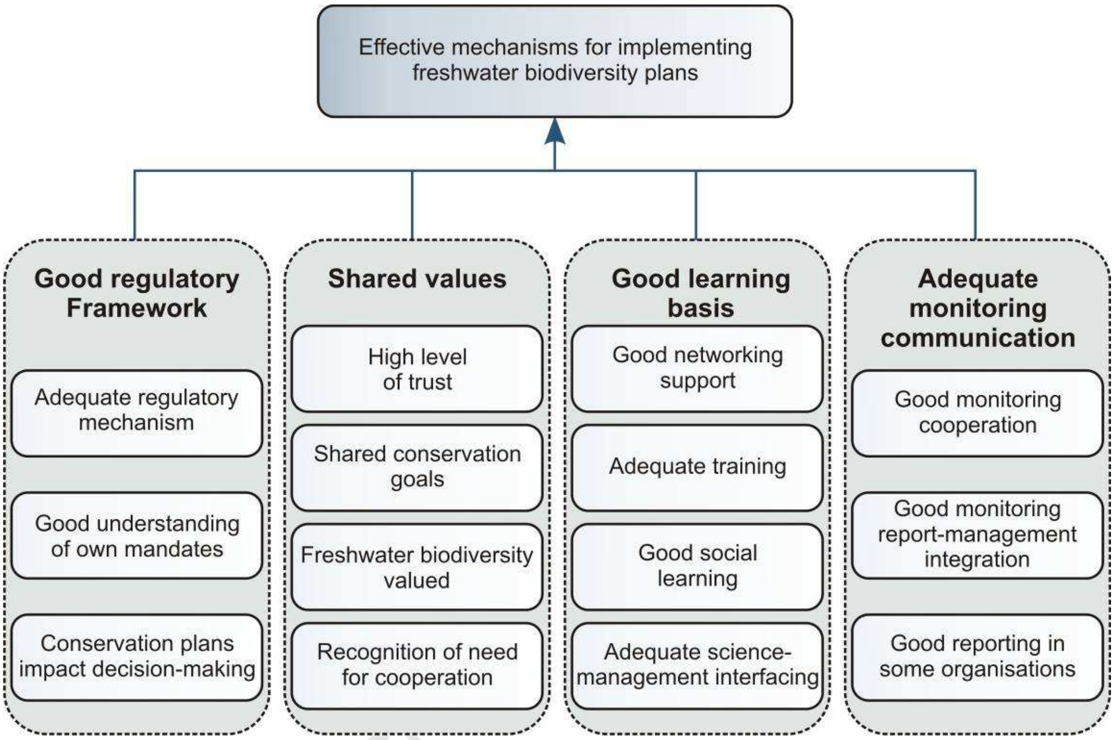

Figure 6.2. Current mechanisms that enable effective implementation of freshwater biodiversity plans.

6.3.1.2. Shared values

A high level of trust exists between individuals in partner organisations (score=3) (Table 6.1), this was illustrated by the ease with which individuals do contact members of partner organisations if they require assistance or just to share information. This has led to mutual problem-solving, especially when meetings are formally organised in settings such as forums, which also accord them the opportunity of building personal relationships with colleagues. Due to the existence of informal networks, other avenues such as formal inter-agency communications were avoided, as a result bureaucratic hurdles were minimised. Participants noted that the real challenge in building a sustained level of trust is to move from trust built between individuals/networks to organisation wide trust.

Table 6.1. Response scores to management effectiveness in the implementation of freshwater biodiversity plans reported in the evaluation of 5 implementing agencies (DWAF, GDACE, NWDACE, NWPARKS, SANBI)

INDICATORS	SD	N	Mean	Average Score	Similarity of scores among Organisations
CONTEXT	0.99				
Clarity of Mandates		4	1.75	Good	Good
Current culture of cooperation		3	2.33	High	Good
Appropriate statutes		5	3.00	High	High
Use of existing statutes		3	1.33	Fair	Good
Capacity to effectively implement regulations		4	0.75	Low	Good
Staff numbers		3	1.67	Good	Good
Staff training		4	2.25	Good	Fair
Equipment		4	2.00	Good	Fair
Ability to influence budget		4	1.50	Fair	Low
Adequacy of budget		4	1.75	Good	Fair
Social learning		4	3.00	High	High
Champion		4	1.00	Fair	High
Networking support		5	2.60	High	Good
Trust		4	3.00	High	High
Freshwater biodiversity value assessment		5	2.20	Good	Fair
PLANNING	0.58				
Participatory target setting		4	1.75	Good	Good
Integration of spatial plans		5	2.00	Good	Fair
Integration between conservation plan and strategic/work plans		5	2.20	Good	Fair
MONITORING	1.00				
Resource inventory		5	1.80	Good	Low
Alignment in Monitoring		5	1.40	Fair	Fair
Cooperation in monitoring		5	1.60	Good	Good
MANAGING	0.55				
Monitoring-reporting-management integration		5	2.60	High	Good
Management plans		5	0.80	Fair	Fair
Science-management interfacing		5	2.20	Good	Good
Impact of conservation plan on decision making		5	1.60	Good	Fair
Reporting		5	1.60	Good	Fair

There was good commonality and performance (score = 2) among partner organisations with regard to shared conservation goals (Table 6.1). During conservation planning, some organisations were involved in setting the biodiversity targets, while other organisations had limited involvement. There seems to be a good understanding of the need to conserve biodiversity and to use resources sustainably, although in some cases this was not clearly reflected in the mandates of the respective organisations. For example it was noted that biodiversity conservation issues generally fall under the jurisdiction of the Department of Environment and Agriculture (DEAT), but the Department of Water Affairs and Forestry (DWAF) dealt with most of the freshwater conservation issues, although in some instances mining was allowed to take place in wetlands.

Freshwater biodiversity is highly valued (score=3) with a good commonality (score= 2) among partner organisations (Fig. 6.2). The conservation of freshwater biodiversity is an integral part of the conservation agenda in the study area. An example was given of the Kgaswane Mountain Reserve, and the Suikerbosrand Nature Reserves that were being managed around their importance as water catchment areas. Traditionally nature reserves were delineated on the basis of their importance for terrestrial biodiversity, but in these examples the carrying capacity of animals was viewed in relation to their impact on freshwater ecosystems. In the study area, where a freshwater biodiversity plan has been developed, it was going to be incorporated into the WMA’s bioregional plan, and was already being used for wetlands restoration in the area by the Working for Wetlands Programme. There was recognition of the need for cooperation with partner institutions, based on the understanding that the mandate for the conservation of freshwater biodiversity is distributed among different sectors. The conservation of freshwater biodiversity required understanding of the threats to the system, the water quality and quantity and the biodiversity therein. To collate all this information required effective cooperation and coordination between implementing agencies, as no single organisation has the mandate to collate all this information. Another critical factor that necessitated cooperation was the trans-boundary nature of freshwater ecosystems, which required coordination beyond geographical and political boundaries for conservation goals to be achieved. In the study area it was noted that although the culture of cooperation appears to be deeply entrenched, it was mostly voluntary and informal and pegged on a few individuals, and can be thrown into disarray because of staff changes at the work place.

6.3.1.3. Good learning basis

It was reported that most organisations provided support for employees to liaise with colleagues in partner organisations (Table 6.1). This has helped in encouraging cooperation because personal relationships are built during such exchanges. Most of the support came in logistical, technological and financial form. But it was also noted that the support provided was often marred by government red tape, where for example provincial authorities have set a limit of 2000km travel distance per month for meetings and fieldwork trips. This was viewed by participants as a hurdle despite the goodwill displayed in encouraging networking between partner organisations.

Adequate staff training exists in some organisations, but the commonality between organisations was not uniform. For example the regional Parks authority reported low staff training (score=1), whereas the department of water affairs (DWAF) rated staff training very highly (score=3). It was, however, acknowledged by all organisations that there was a tendency for increased staff turnover, once staff have received higher training, due to better job offers from elsewhere. The training provided to staff in some organisations was also reported as being too generic and not adequately equipping staff with the technical skills required.

Social learning is taking place and was rated highly (score=3) with high commonality between organisations (Table 6.1). The existence of wetlands forums was particularly noted as a major contributor to social learning. For example an engineer in DWAF reported that he first learnt about the importance of peat wetlands when he attended a wetland forum. The existence of social learning was however, irregular and not well planned. It would make a big difference if social learning were entrenched into organisational functions. A surprising result from this evaluation was the perception that there exists adequate interfacing between science and management. It was however noted that this interfacing only worked up to mid level management.

6.3.1.4. Adequate monitoring and communication

There was a good level (score=2) of cooperation between organisations in biodiversity monitoring (Fig. 6.2). The cooperation was mostly achieved through data-sharing between partner organisations. Cooperation in monitoring across political boundaries also exists, but very informally. The only hindrance to effective monitoring was associated with duplication of effort, where in many cases different organisations monitor the same entities. For example the DWAF, DEAT and other local authorities all monitor water quality in the region. This was as a result of poor coordination between partner organisations in the region. The worst-case scenario presented was of a river named the Blesbokspruit (Gauteng Province), where nine organisations were involved in monitoring the same river. Cooperation in monitoring is also informal, and because of busy schedules individuals who should be coordinating monitoring programs with partner organisations are often too busy monitoring for their own organisations.

At the provincial level of administration there is good integration between monitoring, reporting, and management. There were also some instances where monitoring was part of an adaptive management circle but this was a rare occurrence. Some organisations like DWAF are more interested in hydrological than biodiversity monitoring, and as a result their contribution to effective biodiversity monitoring data is limited. The effectiveness of reporting was acknowledged by some organisations, but not by others.

6.3.2 Barriers to effective implementation of freshwater biodiversity priorities

6.3.2.1. Misaligned strategies and inadequate implementation of conservation plans

It was found that there was very little understanding of the mandates of partner organisations even though participants understood the mandates of their parent organisations. The lack of understanding of partner mandates has affected inter-agency cooperation because the potential channels of collaboration and cooperation are not well defined. In some cases mandates overlap between organisations, and individuals are aware of this but there is no one to champion cooperation between the different agencies.

There was wide disparity in the integration of conservation plans with strategic organisational or individual work plans (Fig. 6.3). Most of the integration was haphazard and at a scale unsuitable for implementation. For example some organisations directly mandated with the protection of freshwater biodiversity have stated the need to integrate conservation plans into their strategic plans. No specific guidelines for integration have been formulated at the levels appropriate for implementation however. This disparity in aligning conservation plans with work/strategic plans was attributed to a lack of vertical policy coherence. All the organisations evaluated also reported a lack of management plans specifically for freshwater ecosystems. The lack of management plans is a major factor contributing to the poor integration of conservation plans.

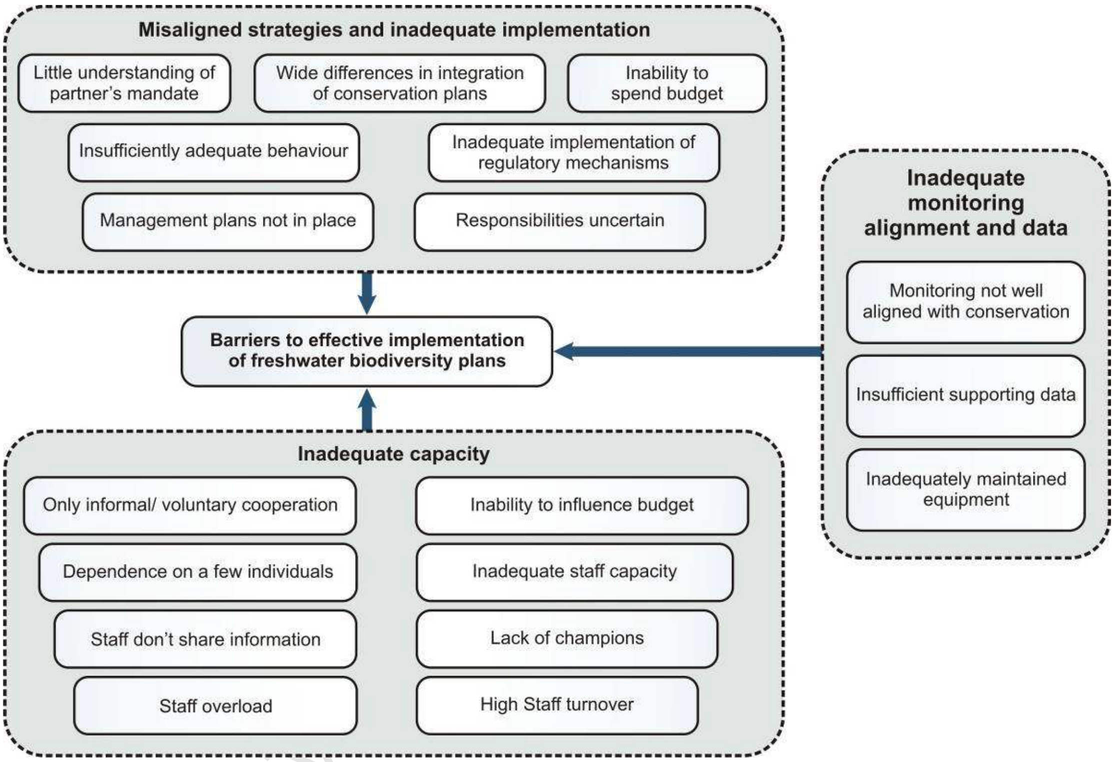

Figure 6.3. Current weaknesses to the effective implementation of freshwater biodiversity plans.

With regard to the integration of spatial biodiversity plans into the broader conservation agenda, participants reported that there was generally a lack deeper-level engagement both within and between organisations. It was therefore reported that the effectiveness of integration was generally lacking in organisations mandated with the protection of freshwater biodiversity. Other factors that were perceived as having led to the poor integration of freshwater spatial plans were a gross lack of stakeholder involvement in the design of the conservation plans, and lack of coordination and accessibility to spatial plans by other sectors like mining, and agriculture. There was also a strong perception that political issues influence decision-making when it came to achieving effective integration of spatial biodiversity plans.

Even though the existence of adequate regulatory frameworks was cited as one of the strengths in most of the organisations, effective implementation of the existing regulatory mechanisms is grossly lacking due to inadequate capacity.

6.3.2.2. Inadequate alignment of monitoring with conservation objectives

There was a perception that although most of the organisations were actively monitoring freshwater resources, there was very poor alignment of the monitoring with conservation objectives (Fig. 6.3). The River Health Program (RHP 2005) was cited as an example of a robust monitoring program but not directly linked to the achievement of specific conservation objectives. Some of the data collated during monitoring was thought to be inappropriate for conservation planning. For example DWAF collates hydrological data but nothing on biodiversity. As a result some participants reported insufficient biodiversity data on some critical habitats in the region for effective management. Some departments have been collating data but the information was not being utilised, either because of the difficulty of integrating different types of data or because the information was deemed irrelevant for their monitoring objectives. Participants also reported the difficulty in accessing data due to absence of a central database. For example in some instances survey data were maintained by the provincial authorities while in some cases field offices were the custodians of such data. The lack of a one-stop was therefore hampering the evaluation of conservation outcomes, and also undermining biodiversity monitoring objectives.

Related to monitoring success is the adequacy and maintenance of equipment. Participants generally agreed that their organisations owned equipment, but there was a very poor record of maintenance. Government procurement processes for purchasing new equipment was very cumbersome. For example in one instance it took 2 years to purchase nets and trays used for monitoring. Furthermore, equipment was lost or in some cases, stolen because when employees left the organisation they took equipment with them.

6.3.2.3. Inadequate capacity

All the organisations reported that staff inadequacy was a major impediment to achieving freshwater conservation goals in the region. The reliance on a few staff with the appropriate expertise and experience was regarded as an impediment to conservation. The inadequacy of staff resulted in increased workloads, because only a few people are able to execute the mandate of the organisation properly. A high staff turnover was also noted by all organisations as negatively impacting on their functioning. Further, because relationships take a long time to build, when staff turnover is very high inter-agency cooperation is impeded and level of trust between individuals in partner organisations is also affected. Closely linked to high staff turnover, staff overload and small number of staff is the lack of champions within organisations. Conservation often relies on a few individuals in the organisations who are passionate about a particular cause and will offer much needed leadership in advancing such a cause. If such charismatic individuals are lost due to staff turnover or become too busy because of work overload, a leadership vacuum is created which will negatively affect freshwater biodiversity conservation.

Most participants also reported a very poor (score = 1.5) ability to control budgets, mainly attributed to poor internal communications. In many cases individuals learn of their budget allocations midway through the financial year, at which point there is not enough time to spend the allocated budgets. This then created an impression of surplus budgets but the money could have been properly spent had it not been a case of poor planning. There was very low commonality (score = 2) between organisations regarding budget adequacy. Whereas DWAF scored budget adequacy very highly, the regional Parks authority scored it very low. Some organisations depended solely on internal funding, while others needed external funding to meet budget deficits.

6.4 Discussion

This study investigated the suitability of current management approaches for implementing freshwater biodiversity priorities and of existing freshwater biodiversity plans for informing catchment management strategies. This was carried out by assessing the management effectiveness of a few institutions directly mandated with the protection of freshwater ecosystems in South Africa. The evaluation gave partner organisations an opportunity for reflection and for learning from each other how to best execute their respective mandates. This was an important outcome of this study that could not be easily reflected in the result, because of the difficulty in ascertaining whether learning took place in the process of the evaluation. The willingness of partner organisations to participate together in a single workshop setting without expressing any reservations was however a good indicator of the recognition of the need to build partnerships.

The emphasis on reflection and self criticism adopted in evaluating management effectiveness was quite pragmatic. This represented a major departure from established management effectiveness approaches used in protected area assessment, which appear to focus more on outcomes (e.g. Hocking et al. 2000, Ervin 2003b). The approach adopted in this study strives to ensure that social learning takes place, rather than primarily seeking to ascertain whether conservation goals are being achieved. Through social learning more emphasis is placed on developing remediation strategies if conservation goals are not being achieved or improving on practices if conservation goals are being achieved. , and inter-institutional transfer of best practices actively takes place (Barber et al. 2004, Hockings et al. 2000). Social learning institutions also help to bridge the gap between conservation assessment and implementation through stakeholder collaboration and strategy development (Knight & Cowling 2007).

The survey indicated that conservation plans were being implemented in cases where such plans existed. But the integration of conservation plans into water resource management strategies is still an ad hoc. In most cases biodiversity plans were being used as tools to facilitate decision-making but were not an integral part of the overall strategic plan. The Biodiversity Act of South Africa (2004), required biodiversity to be incorporated into spatial development initiatives (SDIs), and integrated development plans (IDPs), but little progress appears to have been achieved in this regard. Incorporating biodiversity issues into the SDI and IDP frameworks at a local scale would have provided an excellent opportunity for understanding the trade-offs between biodiversity conservation and development (Wynberg 2002). The difficulty in strategically incorporating biodiversity into these frameworks could partly be attributed to the overall design of the conservation plans, which do not incorporate the socio-political dimension to biodiversity conservation (Knight & Cowling 2007). A lack of understanding of the socio-political landscape in biodiversity planning often results in plans that are completely incompatible with developmental and economic needs of an area. A key requirement for effective implementation of biodiversity priorities is therefore an understanding of the potential trade-offs that might arise during the implementation phase of the biodiversity plan. This could be achieved by ensuring that biodiversity assessment incorporates such issues as landowner willness, resources available to implementing agencies, cost of land for conservation and the suitability of policy frameworks (Knight & Cowling 2007). In this way uncertainties associated with implementing biodiversity plans can be predicted and remedial strategies developed accordingly.

Inter-agency cooperation is critical for the management of water resources, where the responsibility for protecting freshwater biodiversity is distributed between different agencies (Mackay & Ashton 2007, Pahl-Wostl et al. 2007). It was therefore encouraging to find that the need for cooperation was recognised by the different agencies as a key to achieving freshwater conservation goals. However, its dependency on a few individuals as champions makes it precarious. The dependency on a few individuals to champion conservation action is a common occurrence, where in many cases individuals as opposed to the quality of the institution are directly credited with the achievement of conservation objectives (Parker et al. 2009). Due to staff turnover, institutions that depend on a few key individuals to cooperate with colleagues in partner institutions stand to lose those linkages once such individuals leave. A strategic approach would be more useful in fostering cooperation between partner institutions by formalising interactions, and ensuring that linkages are set up at all levels ranging from senior management to field operations. In this way staff turnover will have minimal impact on inter-agency cooperation as the whole process is institutionalised.

The establishment of catchment management agencies (CMA) presents both opportunities and challenges to water resource management in South Africa. For a long time water resource models did not incorporate biodiversity into their management strategies, but the establishment of catchment management agencies (CMAs) is an opportunity to reverse this trend. Even though most CMAs in South Africa have not yet been established, lessons learnt from evaluations of institutions mandated with the management of water resources offers an excellent opportunity to explore ways in which biodiversity issues can be incorporated into catchment management strategies (CMS). Prior to the advent of CMAs different models of management were being used to exploit and protect natural resources (Rogers et al. 2000). The challenge for CMAs is reconciling the divergent management styles. For example the mandate for resource exploitation and protection were performed by two different agencies, while the CMAs are charged with both protection and exploitation of water resources. A major dilemma for CMAs arises because the management scales for water resource protection and exploitation vary, where catchments demarcate water protection boundaries, while water supply is bound by administrative boundaries e.g. Municipalities (Pollard & du Toit 2007). This assessment has set a good example where a deliberate effort was made to bring different agencies together to explore ways of how each can make a unique contribute to the broader goal of biodiversity protection. CMAs could also adopt a similar model for their operations based on working with partners in the different sectors and management jurisdictions in executing their mandates.

The Way forward for systematic conservation planning is to focus on conservation planning products that are strategically geared towards implementation. That requires continuous engagement of stakeholders at all stages of the conservation planning process, right from the biodiversity assessments to implementation. Conservation planning products that explicitly address stakeholder needs are more likely to be successfully implemented because such a process will confer ownership of the conservation plan to the stakeholders and build their capacity. Conservation plans also need to be designed to carry a unified message for ease of interpretation. This calls for an integrated approach to conservation planning, where biodiversity priority areas of different ecosystems are carefully aligned to convey a unified message. Approaches to conservation planning that overlook this challenge are set to face enormous implementation pitfalls. This is because conservation managers who are tasked with biodiversity protection need robust tools that will facilitate their decision-making processes, and presenting them with multiple products is not going to make their work easier. The need for an integrated approach to conservation planning is consistent with (Knight & Cowling 2007, Sowa et al. 2007) call to embrace opportunism, where in many cases day-to-day decisions by conservation managers on development applications hold the key to whether for example, an important wetland is converted into a golf course or not.

CHAPTER 7. SYNTHESIS

This thesis investigated ways to improve freshwater conservation planning by developing tools and approaches for mainstreaming freshwater biodiversity into the conservation agenda and other water management sectors in South Africa. To understand the trends and the prevalent gaps in freshwater conservation planning, a focused review of the systematic conservation planning literature was undertaken, and the outcome of the review led to the formulation of the questions this thesis attempted to address. The key issues that were addressed broadly involved: -

- Developing and testing a framework for automating wetland classification for South Africa with the aim of improving the incorporation of wetlands into conservation planning frameworks (Chapter 3)
- Demonstrating how to effectively plan for freshwater biodiversity persistence in the Cape Floristic Region (Chapter 4)
- Testing an approach for integrating freshwater and terrestrial biodiversity priorities to improve the implementation of conservation action (Chapter 5)
- Assessing the management effectiveness of conservation agencies mandated with the protection of freshwater resources, to determine the efficacy of implementing conservation action in freshwater priority areas (Chapter 6)

In this final chapter, the key questions raised in chapter 1 are revisited and the extent to which this thesis managed to address them are discussed. Where necessary recommendations for future research are proposed.

7.1 Key findings and recommendations

1. To what extent have freshwater ecosystems been incorporated in conservation planning? (Chapter 2)

Even though systematic conservation planning has been widely acknowledged as a prerequisite for developing effective conservation strategies, its application in the freshwater realm is still faced with a lot of challenges. Most of the tools and approaches of systematic conservation planning were pioneered in terrestrial and marine ecosystems, and in many cases do not address the unique challenges associated with freshwater ecosystems, even though the basic principles are applicable to freshwater situations (Dunn 2003, Nel et al. 2008). Some of the principles have started to be implemented in the freshwater context, but progress has been slow so far and some major challenges are still to be overcome. Over the last two decades (1987- 2006), systematic conservation planning techniques started to be widely applied in solving conservation challenges, and their usage in the terrestrial realm grew exponentially (Chapter 2). In this period more than 70% of conservation planning studies focused on terrestrial biodiversity, 20% included some aspects of freshwater biodiversity, and only 5% were designed exclusively for freshwater biodiversity (Chapter 2). There was also a noticeable inconsistency in the measures undertaken to ensure freshwater biodiversity persistence, even though biodiversity persistence remains a key goal of systematic conservation planning (Margules & Pressey 2000, Nel et al. 2008).

It has been only in the last 2- 3 years that significant progress in freshwater conservation planning started to materialise, through the publications of some key papers that boldly attempted to address some of the challenges to freshwater conservation planning (Chapter 2). Issues of connectivity, assessing ecological integrity, freshwater biodiversity surrogates, and incorporating important freshwater ecological processes have been at the core of the failure to adapt systematic conservation planning principles and approaches in freshwater ecosystems, but progress is starting to be achieved in these areas (Chapter 2, Nel et al. 2008). There are however, some challenges that have not yet been addressed more succinctly, such as the integration of biodiversity priorities across the different realms of freshwater, terrestrial and marine ecosystems in order to manage them more effectively (Abell et al. 2002, Sloane et al. 2007, Nel et al. 2008, Tallis et al. 2008). Freshwater conservation planning, has also largely focused on rivers, with minimum incorporation of other wetland types. This can partly be attributed to the challenges of mapping wetlands more effectively, and the fact that wetland ecosystems occur at a transition zone between terrestrial and freshwater ecosystems, where in some cases they have been treated as a terrestrial vegetation type in conservation planning (e.g. Lombard et al. 1997). Generally, the uptake of universally applicable freshwater conservation planning tools and approaches is still relatively slow. There is a need to therefore consolidate the established techniques and apply them in different settings to demonstrate their effectiveness in addressing the challenges to freshwater conservation planning.

2. How can the diversity of wetland types be incorporated in freshwater conservation planning? (Chapter 3)

Wetland classification is important for developing effective conservation strategies, because different wetland types support different biological communities and provide different ecosystem functions and benefits (Hauer & Smith 1998, Standa & Ehrenfeld 2008). Due to the large datasets required in conservation planning, wetland classification in most cases is not effected due to the difficulty in automating proposed classification frameworks. In Chapter 3 it was shown that a simple automation framework based on GIS could be used to accurately implement wetland classification for a large population of wetlands mapped at various scales. The accuracy of the classification varied as the level increased, with coarser levels more accurately classified than finer ones. This framework has major implications for systematic conservation planning, where assessments have often relied on surrogate or incomplete datasets to identify priority areas. The broad wetland classification will enable targets to be set for different wetland types to achieve representation in conservation planning (Ausseil et al. 2007, Chapter 4).

The wetland classification was based entirely on the landscape setting as the key discriminator of wetland types. But to comprehensively reflect the uniqueness of each wetland type, other descriptors such as water source, hydrology and hydrodynamics need to be taken into consideration (Mitsch & Gosselink 2000). An attempt was made to incorporate the hydrological aspect of wetlands in the classification but this did not yield accurate results (Chapter 3). The hierarchical approach enabled wetland classification to be undertaken at various independent levels that gave rise to unique wetland types (Chapter 3). This hierarchical approach brings flexibility to the wetland classification process, so that the level to which wetlands can be classified will depend on the comprehensiveness of the available datasets and the objectives of the classification.

3. How can we effectively plan for freshwater biodiversity persistence? A case study from the Cape Floristic Region, South Africa. (Chapter 4)

This chapter demonstrated how to effectively plan for freshwater biodiversity persistence by jointly incorporating the different freshwater ecosystems, using biodiversity surrogates and exploring the trade-offs between freshwater and terrestrial biodiversity conservation goals.

One of the key lessons learnt in this study is the apparent trade-offs that exist between biodiversity protection and resource utilisation. In South Africa the National Water Act (1998), stipulates that water resources be classified and managed accordingly. But due to the varying objectives of water supply and biodiversity conservation, the classification in some cases results in a trade-off. For example, from a water-supply perspective a river maintained in class D with moderate water quality can achieve water-supply objectives, whereas the same river may not achieve conservation goals if its ecological functions have been disrupted, resulting in a trade-off between biodiversity conservation and utilisation (Dudgeon et al. 2006, Nel 2008). Another aspect of this trade-off is in terms of representation, where every single river in good condition is important for achieving water supply objectives; the same does not apply to biodiversity conservation. In this study for example 35% of the rivers were classified as A or B, but only 8% of those rivers contributed towards target achievement (Chapter 4). This is because these rivers were not diverse in terms of river types, and most of them were therefore not selected as freshwater priorities despite being in good condition (Chapter 4). The trade-offs between biodiversity representation and meeting water supply objectives are very difficult to overcome because both issues are critical in water resource management.

The second trade-off is between freshwater and terrestrial conservation objectives. Even though the overlap between freshwater and terrestrial priority areas was reasonable (28%), a major disparity was found in the protection of individual freshwater biodiversity features. This was supported by the general assertion in the literature that freshwater ecosystems are mostly included in conservation planning frameworks because of their importance for terrestrial biodiversity (Chapter 2, Madsen et al. 1998). This was clearly evident in the CFR where the terrestrial conservation plan used river networks as corridors for connecting terrestrial priority areas (Cowling et al. 2003). Although the CFR study took into account fish species, measures for their freshwater biodiversity persistence were not well defined, which may explain the reasons for failing to achieve fish conservation targets (Chapter 4).

4. *How can freshwater and terrestrial biodiversity be optimally integrated in conservation planning? (Chapter 5)*

The attempt to integrate different ecosystem types in conservation planning is fraught with challenges especially between freshwater and terrestrial ecosystems. The longitudinal nature of river systems defies the use of terrestrial planning units, and instream ecological integrity is influenced by the wider catchment (Amis et al. 2007). These differences have made the integration of freshwater and terrestrial biodiversity in conservation planning particularly challenging (Chapter 5). In chapter 5 a protocol was proposed for integrating freshwater and terrestrial biodiversity priorities in conservation planning, with the aid of a decision support system (DSS). The step-wise approach involved the separate assessment of freshwater priority areas, and then the use of the outcome to influence the selection of terrestrial priority areas (Chapter 5). This allowed the preferential selection of areas where freshwater and terrestrial priorities overlap.

The key findings show that both freshwater and terrestrial biodiversity targets could be optimally achieved, where the overlap in priority areas for both systems improved from 23% to 47%. Target achievement for freshwater biodiversity improved by 10% when they were used to drive the selection of terrestrial priority areas. It was shown that using freshwater biodiversity to plan for both systems is a more efficient approach than basing conservation decisions on terrestrial priorities with the assumption that freshwater biodiversity will be protected too. The findings here are supported by several studies that show terrestrial conservation strategies as inadequate for securing freshwater biodiversity (Keith 2000, Roux et al. 2008). Catchments are the principal planning and management units of freshwater biodiversity, because they comprise the main area that drains the freshwater ecosystems. And because catchments are comprised of terrestrial biodiversity, it is possible achieve terrestrial conservation goals when planning for freshwater ecosystems (Abell et al. 2007, Chapter 5).

5. Are there enabling mechanisms for the effective implementation of freshwater biodiversity priorities in South Africa? (Chapter 6)

A large disconnect exists between conservation planners and potential implementers. In this chapter it was argued that to develop effective freshwater conservation strategies with potential for implementation requires a good understanding of the strengths and weaknesses of the current mechanisms for the protection of freshwater ecosystems in the study region. Systematic conservation planning is a continuum from biodiversity assessment to implementation, but rarely are implementation challenges addressed as part of this continuum (Knight et al. 2006). More often the only conservation planning product is an elaborate map showing where biodiversity priorities are located in the landscape, but with no accompanying guidelines as to how those sites could be secured. Such conservation plans become redundant because of a lack of clear a strategy. This predicament could be partly attributed to the wide divide that exists between scientists who develop the conservation plans and practitioners who are the implementers (Prendagast 1999, Knight et al. 2006).

A key lesson learnt from the analyses of the perceptions/opinions of key conservation managers is that freshwater conservation plans need to be designed to overcome specific implementation challenges, and should be relevant to management (Holling et al. 1995). This requires flexibility (Abellan et al. 2005) to incorporate potential implementation challenges more succinctly into the process of conservation planning. For example the inability of conservation managers to interpret and use conservation planning products for decision making (Pullin & Knight 2005) requires that conservation plans be simplified to allow for ease of interpretation by conservation managers. In cases where the lack of stakeholder buy-in or conflicts between stakeholders might impede implementation, requires that stakeholder involvement is made a key priority throughout the conservation planning process.

Implementing freshwater priorities should be taken as a learning experience in an adaptive management framework (Pahl-Wostl 2002, Knight et al. 2006). In this way conservation plans are viewed as dynamic tools for decision making as opposed to static products. Making conservation plans part of an adaptive management cycle will enable periodic review to determine whether conservation goals are being achieved or not, it will also enable the trade-offs between biodiversity protection and other uses to be accounted for.

7.2 Recommendations for future research

There is a need to extend the framework developed for integrating different ecosystems to include marine systems, in order to understand the interactions between land and sea (Sloane et al. 2007). This thesis focused on the integration of freshwater and terrestrial biodiversity assessments; the logical thing to do now is to explore how these systems link up to marine ecosystems. Integrating the different terrestrial, freshwater and marine ecosystems will help in understanding threats to marine systems emanating from the terrestrial landscape (Tallis et al. 2008).

Although this thesis addressed some pertinent challenges in planning for freshwater biodiversity persistence (Chapter 5), it did not address the challenges posed to freshwater ecosystems by climate change. Freshwater ecosystems are expected to be severely impacted by climate change, with increased temperatures and changes in moisture regimes impact ing on species distribution (IPCC 2008). At present freshwater conservation planning approaches do not incorporate measures of climate change vulnerability when identifying freshwater biodiversity priorities (Abell et al. 2002). Although healthy catchments and free-flowing rivers are likely to be resilient to climate change, other measures of climate change adaptation and mitigation need to be built into approaches for conservation planning. There is a need to understand which freshwater biotas are most vulnerable to climate change and incorporate this information when setting targets during conservation planning.

An issue that has become increasingly important but not properly dealt with in systematic conservation planning studies, including this thesis is that of environmental flows. Most systematic conservation plans are concerned with biodiversity assessments as the key determinant of conservation priority areas, but for persistence of instream biodiversity there is a need to maintain the minimum flows required for aquatic habitats. Building the linkage between environmental flow allocations and the process of systematic conservation planning is therefore an issue that is highly recommended for future research. This has become even more pertinent in light of climate change threats to freshwater ecosystems, and the potential of using environmental flows as a strategy for adapting to climate change.

There is a need to develop more effective approaches for incorporating freshwater connectivity, and upstream threats in conservation planning. At present connectivity issues in freshwater conservation planning are only incorporated during the design phase of the planning (Chapter 4), but this has a potential to affect the spatial efficiency of the priority areas. Lateral connectivity was the only aspect of freshwater connectivity that was addressed in this thesis, when catchments were used as the planning units and priority river reaches were buffered (Chapter 4). Developing measures that take connectivity into account as priority areas are identified would be a more objective approach to freshwater conservation planning.
There is a need to adopt multidisciplinary approach to freshwater conservation planning, by integrating socioeconomic and political factors when identifying priority areas. This is because most of the major determinants of effective implementation are non-scientific

(Chapter 6). Freshwater conservation planning therefore needs to take a pragmatic approach by incorporating more than just biodiversity data when identifying freshwater priority areas. This need has long been recognised in terrestrial and marine conservation planning (Pressey et al. 1997, Lundquist & Granek 2005, Knight & Cowling 2008), but its application in freshwater conservation planning is still relatively unexplored, despite the fact that freshwater ecosystems present enormous challenges to their management because of the complex political processes involved (Dollar et al. 2007, Pahl-Wostl et al. 2007).

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APPENDICES

Appendix I: SANBI 2009. The latest version of the proposed wetland classification system for South Africa

CONNECTIVITY TO OPEN OCEAN	***ECOREGION***	***LANDSCAPE SETTING***	***FLUVIAL INTEGRATION***	***LANDFORM & HYDROLOGY***		***DRAINAGE***
		A	**B**	**A**	**B**	**C**
INLAND	DWAF Level I Ecoregions	SLOPE	STRONGLY INTEGRATED	Channel (river)	Mountain headwater	
					Mountain stream	
					Transitional	
				Depression		Dammed
			WEAKLY INTEGRATED	Hillslope seep		With channelled outflow
						Without channelled outflow
				Depression		Dammed
			ISOLATED	Depression		(not applicable)
		VALLEY FLOOR	STRONGLY INTEGRATED	Channel (river)	Transitional	
					Upper foothill	
					Lower foothill	
					Lowland	
				Channelled valley-bottom wetland	Valley-bottom depression	Exorheic
						Endorheic
					Valley-bottom flat	
				Unchannelled valley-bottom wetland	Valley-bottom	Exorheic

CONNECTIVITY TO OPEN OCEAN	***ECOREGION***	***LANDSCAPE SETTING***	***FLUVIAL INTEGRATION***	***LANDFORM & HYDROLOGY***		***DRAINAGE***
		A	**B**	**A**	**B**	**C**
					depression	Endorheic
					Valley-bottom flat	
				Floodplain wetland	Meander cut-off	
					Backwater depression	Exorheic
						Endorheic
					Floodplain flat	
				Depression		Exorheic
						Endorheic
						Dammed
			WEAKLY INTEGRATED	Depression		Exorheic
						Endorheic
						Dammed
			ISOLATED	Depression		Exorheic
						Endorheic
						Dammed
		PLAIN	STRONGLY INTEGRATED	Channel (river)	Lowland	
				Floodplain wetland	Meander cut-off	
					Backwater depression	Exorheic
						Endorheic
					Floodplain flat	
				Unchannelled valley-bottom wetland	Valley-bottom depression	Exorheic
						Endorheic
					Valley-bottom flat	
				Depression		Exorheic
						Endorheic
				Flat		

CONNECTIVITY TO OPEN OCEAN	*ECOREGION*	*LANDSCAPE SETTING*	*FLUVIAL INTEGRATION*	*LANDFORM & HYDROLOGY*		*DRAINAGE*
		A	**B**	**A**	**B**	**C**
			WEAKLY INTEGRATED	Depression		Exorheic
						Endorheic
				Flat		
			ISOLATED	Depression		Exorheic
						Endorheic
				Flat		
		BENCH (HILLTOP / SADDLE / SHELF)	WEAKLY INTEGRATED	Depression		Exorheic
						Endorheic
				Flat		
				Valleyhead seep		
			ISOLATED	Depression		Exorheic
						Endorheic
				Flat		

Latest version of the Marine wetland classification system modified from Ewert-Smith 2006						
LEVEL 1: SYSTEM	**LEVEL 2: INSHORE BIOREGION**	**LEVEL 3: SUBSYSTEM**	**LEVEL 4: HYDROGEOMORPHIC UNIT**			
			A	**B**	**C**	**D**
Connectivity to open ocean	*Biogeography*	*Wave exposure*				
MARINE	Inshore Bioregions used in NSBA for Marine Ecosytems	Exposed Coast				
		Sheltered Coast (Embayment)				

Latest version of the estuarine wetland classification system. Modified from Ewart-Smith 2006

<table>
<tr><th rowspan="2">LEVEL 1: SYSTEM</th><th rowspan="2">LEVEL 2: BIOGEOGRAPHIC ZONE</th><th rowspan="2">LEVEL 3: SUBSYSTEM</th><th colspan="4">LEVEL 4: HYDROGEOMORPHIC UNIT</th></tr>
<tr><th>A</th><th>B</th><th>C</th><th>D</th></tr>
<tr><td>Connectivity to open ocean</td><td>Biogeography</td><td>Periodicity of connection</td><td colspan="4">Landform and hydrodynamics</td></tr>
<tr><td rowspan="5">ESTUARINE</td><td rowspan="5">Biogeographic Zones used in NSBA for Estuarine Ecosystems</td><td rowspan="3">Permanently Open</td><td>Estuarine Bay</td><td></td><td></td><td></td></tr>
<tr><td>Open Estuary</td><td></td><td></td><td></td></tr>
<tr><td>River Mouth</td><td></td><td></td><td></td></tr>
<tr><td rowspan="2">Temporarily Open/Closed</td><td>Estuarine Lake</td><td></td><td></td><td></td></tr>
<tr><td>Closed Estuary</td><td></td><td></td><td></td></tr>
</table>

Appendix II: A reflective assessment process for promoting multi-agency cooperation: Towards achieving cross-sector policy objectives for conserving freshwater ecosystems (Roux et al. 2009)

No.	Issues, indicators and criteria	
1	***CONTEXT Issue 1 : Clarity of respective mandates***	
	Do you have a clear understanding of the mandate of your and relevant other organisations?	
	Do not have a clear understanding of my organisation's mandate	0
	Understand my organisation's mandate	1
	Understand my organisation's mandate and have some understanding of the mandates of partner organisations	2
	Have a clear understanding of the complementarity between the mandates of my organisation and those of partner organisations	3
2	***CONTEXT Issue 2 : Current culture of cooperation***	
	What is the current culture of cooperation between you and your partner organisation/s regarding various aspects of natural resource monitoring and management?	
	Virtually no co-operation takes place between you and your partner organisation/s	0
	There is informal co-operation between you and your partner organisation/s on an irregular basis	1
	There is regular but not formalised co-operation between you and your partner organisation/s	2
	There is regular AND formalised co-operation between you and your partner organisation/s regarding the conservation of freshwater biodiversity	3
3	***CONTEXT Issue 3 : Appropriate statutes***	
	Are legal mechanisms in place for the conservation of freshwater ecosystems?	
	There are no legal mechanisms for the conservation of freshwater biodiversity	0
	Relevant parties are in agreement on the need for legal mechanisms for the conservation of freshwater biodiversity but the process of drawing up legal mechanisms has begun but is still incomplete	1

No.	Issues, indicators and criteria	
	Sufficient legal mechanisms for the conservation of freshwater biodiversity are in place	2
	Sufficient legal mechanisms for the conservation of freshwater biodiversity are in place and priority freshwater ecosystems are explicitly and effectively linked to these mechanisms	3
4	***CONTEXT Issue 4 : Use of existing statutes***	
	Are agricultural, mining and industrial uses of land and water being regulated?	
	No mechanisms exist for regulating agricultural, mining and industrial uses of land and water	0
	Mechanisms for regulating agricultural, mining and industrial uses of land and water exist but there are major problems in implementing them effectively	1
	Mechanisms for regulating agricultural, mining and industrial uses of land and water exist with few problems in implementing them effectively	2
	Mechanisms for regulating agricultural and industrial uses of land and water exist and are implemented effectively	3
5	***CONTEXT Issue 5 : Capacity to effectively implement regulations***	
	Can staff implement agricultural, mining and industrial water and land use regulations effectively?	
	The staff have no effective capacity/resources to implement agricultural, mining and industrial water and land use regulations effectively	0
	There are major deficiencies in staff capacity/resources to implement agricultural, mining and industrial water and land use regulations effectively (for example, high staff turnover and insufficient budget)	1
	Staff have reasonable capacity to implement agricultural, mining and industrial water and land use regulations effectively, but some deficiencies remain	2
	Staff have excellent capacity/resources to implement agricultural, mining and industrial water and land use regulations effectively	3
6	***CONTEXT Issue 6 : Staff numbers***	
	Do you have sufficient staff and all the required skills (in your organisation) to effectively conserve freshwater biodiversity? Consider whether you have the following skills or capacity: Fish biologist; aquatic invertebrate specialist; water quality specialist; hydrologist; botanist; geomorphologist; wetland ecologist; GIS specialist; conservation planner	
	There are no staff	0
	Staff numbers are inadequate and staff are unqualified	1
	Staff numbers are below optimum level, but staff are well qualified	2
	Staff numbers are adequate and staff are qualified to undertake freshwater conservation planning	3

No.	Issues, indicators and criteria	
7	***CONTEXT Issue 7 : Staff training***	
	Do staff receive appropriate training?	
	Staff undergo no training	0
	Staff receive generic training only	1
	Staff receive only theoretical (e.g. conferences or courses) or practical training (e.g. fieldwork and application of methodologies) on freshwater conservation	2
	Staff receive both practical and theoretical training on a regular basis (at least twice per year)	3
8	***CONTEXT Issue 8 : Equipment***	
	Do you have sufficient equipment to effectively conserve freshwater biodiversity?	
	There are limited or no equipment and facilities	0
	There are some equipment and facilities, but these are wholly inadequate	1
	There are equipment and facilities, but still some major gaps that constrain management	2
	There are adequate equipment and facilities	3
9	***CONTEXT Issue 9 : Ability to influence budget***	
	Do you know your available budget for freshwater conservation and can you influence it?	
	The size of the budget is not made known and it is impossible to influence it	0
	The size of the budget is only made known after the start of the financial year and cannot be influenced	1
	The size of the budget is made known at the beginning of the financial year and can be influenced to a certain extent	2
	The size of the budget is known at least one year in advance and can be influenced prior to allocation	3
10	***CONTEXT Issue 10 : Adequacy of budget***	
	Do you have an adequate budget for the implementation of your freshwater objectives?	
	There is no secure budget for freshwater conservation and management for your organisation, which is wholly reliant on external funding	0
	There is very little secure funding, and your organisation cannot implement freshwater objectives without external funding	1
	There is a reasonably secure core budget	2

No.	Issues, indicators and criteria	
	There is a secure budget for freshwater conservation planning and implementation on a multi-year cycle	3
11	***CONTEXT Issue 11 : Social learning***	
	Are you learning with your partners? (inter-organisation)	
	No social learning takes place between partners	0
	Limited and mostly ad hoc social learning takes place, either in the field or during meetings	1
	Planned events take place occasionally during which social learning includes both theory and practice	2
	Regular and planned events take place during which social learning includes both theory (e.g. conceptual discussions) and practice (e.g. fieldwork)	3
12	***CONTEXT Issue 12 : Existence of a champion***	
	Do you have a coordination champion for freshwater conservation (individual or core group)? (inter-organisation)	
	No individual or core group has emerged as a champion for partner cooperation	0
	Partners meet on an ad hoc basis and without the direction of a champion	1
	A champion coordinates some relevant activities on an annual basis	2
	A champion is accepted by all partners and he/she actively facilitates coordinated action and co-learning and this role is supported by your organisation	3
13	***CONTEXT Issue 13 : Networking support***	
	Does your organisation provide support for networking with partners? (intra-organisation)	
	Your organisation provides no support for external networking	0
	Your organisation provides limited support for networking with immediate partner organisations	1
	Your organisation provides logistical, technological and financial support for networking with partner organisations	2
	Your organisation actively promotes and provides logistical, technological and financial support for networking with partner organisations as well as with external, but relevant knowledge sources (e.g. universities, conferences)	3
14	***CONTEXT Issue 14 : Trust***	
	Is there a healthy level of trust between partners? (inter-organisation)	
	You do not know who your counterparts are or have virtually no contact with them	0

No.	Issues, indicators and criteria	
	Limited and ad hoc interaction between counterparts takes place, including some discussion of freshwater conservation issues	1
	You feel comfortable to ask your counterpart(s) for assistance in achieving your mandate	2
	It comes naturally to phone your counterparts and freely discuss issues related to freshwater conservation, including mutual problem solving across organisational boundaries	3
15	***CONTEXT Issue 15 : Perceived value of freshwater biodiversity***	
	Is freshwater biodiversity valued?	
	The need to conserve freshwater ecosystems is rather invisible or obscure within the portfolio of organisational priorities	0
	Your organisation shows significant intent to conserve freshwater ecosystems, but action generally lacks	1
	The need to conserve freshwater ecosystems is widely understood in your organisation and some success stories exist	2
	Conservation of freshwater biodiversity features as a high priority on management and policy agendas and this is reflected in strong support for and active initiatives to understand, identify and conserve this biodiversity	3
16	***PLANNING Issue 16 : Participatory target setting***	
	Is there a social mechanism for sharing complementarity of target allocation, keeping in mind land use planning, high value conservation areas, connecting gradients, recognition of natural disturbances?	
	No targets have been set	0
	Conservation targets have been set but we were not involved	1
	We were involved in target setting but inclusive ownership lacks	2
	Target setting is an ongoing and fully participatory process, including involvement from organisations outside our domain of responsibility to ensure that large-scale ecosystem processes are covered	3
17	***PLANNING Issue 17 : Integration of spatial plans***	
	Are different forms of spatial planning in your region well aligned? (inter-organisation)	
	Each form of spatial planning takes place in isolation	0
	Some sharing of data and products takes place between planning initiatives	1
	Sharing is common and some integration takes place	2

No.	Issues, indicators and criteria	
	Full integration takes place between spatial planning, including freshwater and terrestrial conservation planning, catchment management planning and spatial development planning	3
18	***PLANNING Issue 18 : Integration between conservation plan and strategic/work plans***	
	Are these priorities reflected in your organisation's strategic plan / work plan?	
	Conservation priorities and actions are not reflected in the strategic plan of the organisation and work plans of individuals	0
	Priorities are reflected in the strategic and work plans but are only partially implemented	1
	Priorities are fully integrated in strategic and work plans and implemented	2
	Priorities are fully integrated in strategic and work plans, implemented and are regularly reviewed	3
19	***MONITORING Issue 19 : Resource inventory***	
	Does your organisation (department) have enough information to manage the area?	
	There is little or no information available on the critical habitats, species and cultural values of the area (province or management area)	0
	Information on the critical habitats, species and cultural values of the area (province or management area) is insufficient to support planning and decision making	1
	Information on the critical habitats, species and cultural values of the area (province or management area) is not quite sufficient to support planning and decision making but necessary survey work is in place	2
	Information on the critical habitats, species and cultural values of the area (province or management area) is sufficient to support planning and decision making and necessary survey work is being maintained	3
20	***MONITORING Issue 20 : Alignment of monitoring***	
	Is monitoring aligned with the achievement of freshwater conservation objectives? (intra-organisation)	
	No relevant monitoring is undertaken by your organisation	0
	Some relevant monitoring is undertaken	1
	Your organisation actively participates in relevant monitoring programme, e.g. the River Health Programme, but does not link results directly to conservation objectives	2
	A monitoring programme that is aligned with freshwater conservation objectives and targets (e.g. the River Health Programme) has official status and is being maintained	3

No.	***Issues, indicators and criteria***	
21	***MONITORING Issue 21 : Cooperation in monitoring***	
	Are monitoring responsibilities shared amongst partners? (inter-organisation)	
	Different monitoring activities take place in isolation; not familiar with partners' monitoring activities	0
	Aware of partners' monitoring activities, but no cooperation is taking place	1
	Some cooperation in monitoring, but information management systems remain independent	2
	There is integrated design (agreed-on indicators) and coordination in monitoring amongst all partners; compatibility of, access to, and transfer of data are well advanced	3
22	***MANAGEMENT Issue 22 : Monitoring-reporting-management integration***	
	Do you have an integrated monitoring, reporting and management system? (intra-organisation)	
	No monitoring or reporting takes place	0
	Monitoring or reporting takes place but not as a linked system	1
	Regular monitoring and reporting take place and are mutually reinforcing	2
	Regular monitoring and reporting activities take place, are mutually reinforcing, linked to conservation targets, and a clear mechanism exists for results to inform management decisions (adaptive management)	3
23	***MANAGEMENT Issue 23 : Management plans***	
	Are there management plans for freshwater conservation areas?	
	There are no management plans for freshwater conservation areas	0
	Some management plans exist or are being developed	1
	Management plans exist for the majority of identified freshwater conservation areas	2
	Each identified freshwater conservation area has a management plan that includes required actions, target objectives, timeframes and responsibilities	3
24	***MANAGEMENT Issue 24 : Science-management interfacing***	
	Is there a science-management link/continuum in place (where some scientists act as managers and some managers act as scientists?)	
	We do not have any scientists	0
	Scientists and managers work completely separately	1

No.	Issues, indicators and criteria	
	Scientists and managers work together to some extent	2
	Scientists and managers actively and constructively influence each other's thinking and actions	3
25	***MANAGEMENT Issue 25 : Impact of conservation plan on decision making***	
	Are land-use decisions and water use allocations in the area made in accordance with specific guidelines based on your conservation plan?	
	Decisions are not made according to any set process or guidelines	0
	Guidelines exist but there is a lack of clarity on whether decisions are made in accordance with them	1
	Guidelines exist and the compliance of decisions to the guidelines are monitored, but a significant number of decisions are not made in accordance with the guidelines	2
	Guidelines exist and the compliance of decisions is monitored and the majority of the decisions are in compliance with the guidelines	3
26	***MANAGEMENT Issue 26 : Reporting***	
	Are regular reports produced on the status of freshwater ecosystems?	
	No reports are produced	0
	Reports are produced irregularly and not based on quantitative data	1
	Reports are produced regularly and are based on quantitative data, but do not show trends over time or relate directly to decision-making	2
	Reports are produced regularly based on consistent indicators which track changes over time and feed directly into decision-making	3