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**ENHANCING THE CONSERVATION OF
FRESHWATER BIODIVERSITY
THROUGH IMPROVED FRESHWATER
CONSERVATION PLANNING TECHNIQUES**

By:

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**Thesis presented for the degree of
Doctor of Philosophy**

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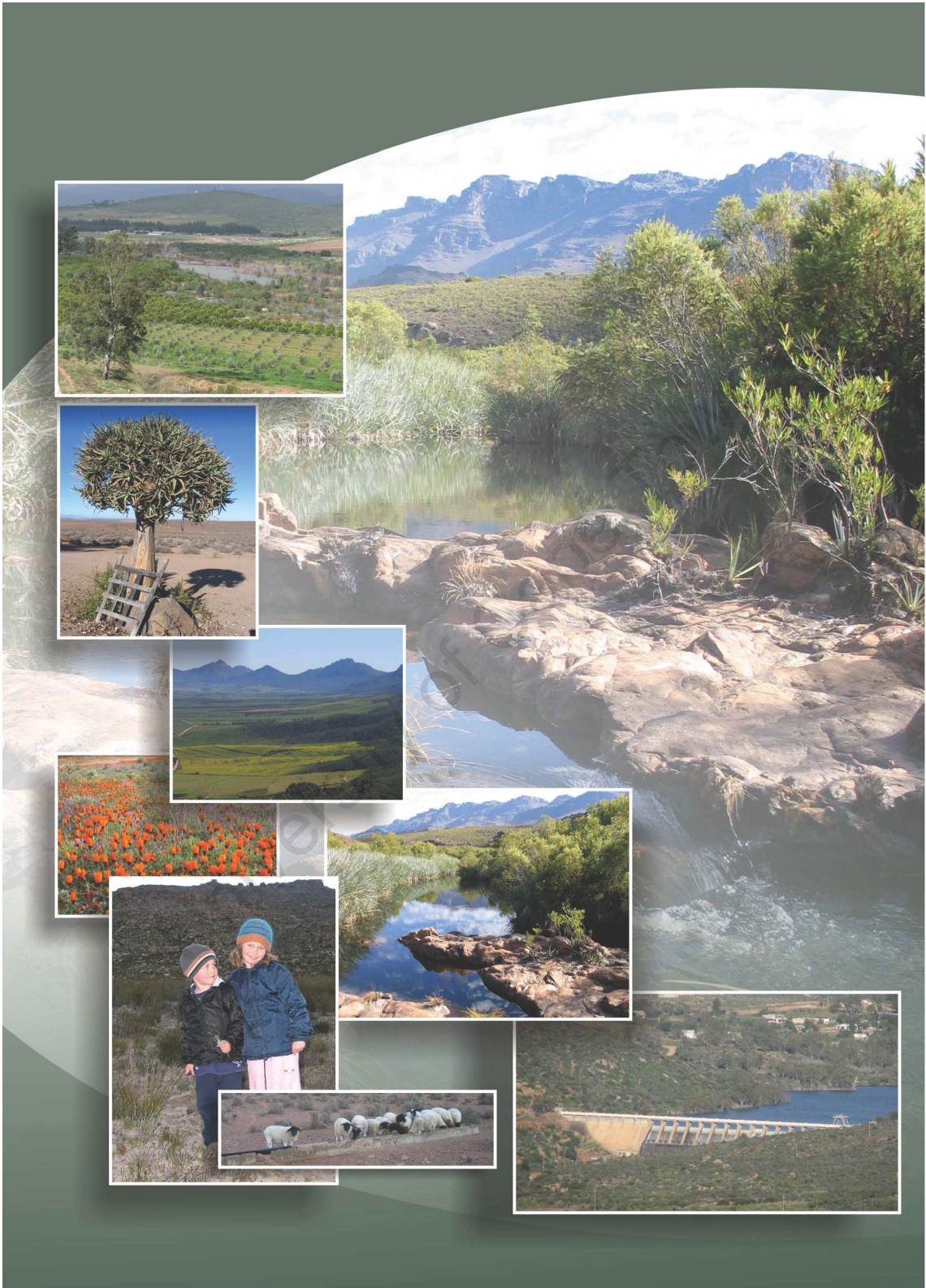
August 2008

For Deon

&

Rachel and Robbie:-

*May your children's children enjoy
the Cederberg rivers and pools as much as we do*



Photographs by Caroline Gelderblom, Jeanne Nel and Deon Nel

Scenes from the Olifants/Doorn case study area in Chapter 6.

DECLARATION

The work presented in this thesis is the sole responsibility of the candidate although four of the chapters have been published or submitted for publication with several co-authors. The candidate is the principle author of all four papers, with the co-authors playing an advisory role either in the initial stages of data collation, or in review of the draft manuscript. Chapter 2 has been accepted by *Aquatic Conservation: Marine and Freshwater Ecosystems*, pending minor revision that has been incorporated into the PhD chapter; Chapter 3 has been published by *Diversity and Distributions*; Chapters 4 and 6 have been submitted to *Biological Conservation* and *Freshwater Biology* respectively. The specific roles of the co-authors are acknowledged at the end of each of the four chapters. This work has not been submitted for a degree at any other university.

Jeanne Lindsay Nel, August 2008

ABSTRACT

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Freshwater ecosystems and biota are among the most endangered in the world. This current situation is even more disturbing when future threats of escalating human demand and global climate change are considered. Urgent measures are therefore needed to conserve freshwater ecosystems and sustain the services they provide. These may take the form of formal protection but also need to include less restrictive mechanisms, such as implementing integrated catchment management and environmental water requirements. Systematic conservation planning provides a strategic and scientifically defensible framework for doing this. Pioneered in the terrestrial realm, uptake of systematic conservation planning for freshwater ecosystems has been slow. While broad principles are applicable, approaches need to be freshwater-specific. The lack of freshwater-specific frameworks and tools is a key factor hampering the application of systematic conservation planning in the freshwater realm. The aim of this thesis was to address this need by developing a suite of frameworks and practical applications for planning in freshwater settings.

The development of a framework for the rapid assessment of river ecosystem endangerment and protection levels provided a common currency for comparing the state of biodiversity across terrestrial and aquatic realms. It showed, for the first time, that the state of river ecosystems in South Africa is dire, far worse than that of terrestrial ecosystems. In addition, river ecosystems have very low levels of representation in protected areas, with many not represented at all. A more optimistic finding was that river systems in protected areas appear to be in a better overall condition than those outside of protected areas, emphasizing the potential of protected areas in conserving freshwater ecosystems.

Currently, however, protected area systems worldwide show significant gaps in their conservation of freshwater biodiversity. A framework was therefore developed for locating and designing protected area systems for the benefit of river biodiversity. Conservation objectives were established for improving river biodiversity pattern and processes in both new and existing protected areas. These included representation of river ecosystems and freshwater fish species, representation of large-scale biodiversity

processes associated with free-flowing rivers and catchment-estuarine linkages, and improving the persistence of river reaches already contained within protected areas. Data were collated in a Geographic Information System (GIS) and a conservation planning algorithm was used as a means of integrating the multiple objectives in a spatially efficient manner. Realistically, protected areas can only play a partial role in overall efforts to conserve freshwater biodiversity and need to be supplemented with other off-reserve conservation strategies. In addition, conservation strategies that focus only on representation of biodiversity in isolated areas are conceptually flawed, especially given the inherent connectivity of freshwater ecosystems.

Such conservation strategies need to be augmented with approaches that address the persistence of freshwater biodiversity. A framework for planning for the persistence of freshwater biodiversity was therefore developed, synthesizing concepts from freshwater ecology and terrestrial conservation planning. When considering issues of persistence, making use of a multiple-use zoning strategy is a practical option because it helps to emphasize that different levels of protection, and hence utilization, can be afforded to different conservation areas. This helps to strengthen the linkages between people and conservation, and aligns more closely with planning categories used by water resource managers and land use planners.

Planning for both representation and persistence should be achieved simultaneously to maximize spatial efficiency. Several methods of planning for representation and persistence were explored. An existing conservation planning algorithm (MARXAN) was adapted for use in freshwater settings through the incorporation of directional connectivity considerations. When using a conservation planning algorithm, the manner in which spatial efficiency between persistence and representation is achieved depends on whether or not a multiple-use zoning strategy will be applied during design. Given the practicalities of multiple-use zoning at local levels of planning, it is recommended that zones should be used in the design phase, rather than merely allocated at the end once the design is complete.

In summary, research and practice in conservation has tended to focus on terrestrial biodiversity; while water resources management has tended to have a more utilitarian focus. It is high time to elevate freshwater biodiversity concerns on the agendas of both these sectors. By developing common conservation frameworks around which the water and conservation sector can engage and debate, this thesis attempts to enhance the integration of freshwater biodiversity concerns into both these sectors.

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CHAPTER 1.
GENERAL INTRODUCTION

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GENERAL INTRODUCTION

Biodiversity conservation is about sustaining the variety of life on Earth, including all ecosystems, biological assemblages, species and populations (Convention on Biological Diversity, <http://www.biodiv.org>). In recent decades, there has been a growing realization that freshwater biodiversity worldwide has been severely impacted by human activities. This is reflected by an expanding evidence base indicating that freshwater ecosystems and their associated biota are among the most endangered in the world (Ricciardi and Rasmussen, 1999; WWF, 2004; Dudgeon et al., 2006). Future prospects for freshwater biodiversity look alarmingly bleak. The indications are that human demand for water will continue to grow at an exponential rate, and it is predicted that 48 % of all people will live in water stressed catchments by 2025 (Zemen et al., 2006). This will result in ever-increasing habitat degradation and loss of biodiversity that is further confounded by the effects of global climate change, which threaten water supplies in many regions of the world (Malmqvist and Rundle, 2002).

Numerous calls have therefore been made for urgent attention to be given to the conservation of freshwater biodiversity (Abell, 2002; Saunders et al., 2002; Dunn, 2003; Dudgeon et al., 2006). Apart from conserving freshwater biodiversity for its own sake, there is also a strong social and economic argument. Maintenance of biodiversity underpins the healthy functioning of freshwater ecosystems, which in turn provide valuable ecosystem services such as potable water, nutrient sequestration, flood regulation, and the provision of exploitable plants and animals (Millennium Assessment, 2005). There is a growing body of evidence linking the impacts on biodiversity (e.g. changes in invertebrate assemblages, presence of key species, magnitude of species richness, and other attributes of communities) to the degradation of ecosystem functioning (Dudgeon et al., 2006). This reduces the ability of freshwater ecosystems to absorb natural and anthropogenic disturbances (Palmer et al., 2008), which ultimately leads to the need for more costly management interventions, such as water purification and flood control. The downward trend of freshwater biodiversity therefore also puts aspects of economy and quality of life at risk, and reduces the

spectrum of socio-economic options available to future generations, particularly in the face of global climate change.

The strong connectivity of freshwater ecosystems is one of the key factors contributing to this widespread degradation (Dudgeon et al., 2006). Established concepts in freshwater ecology emphasize the highly dynamic, heterogeneous and interconnected nature of freshwater ecosystems. We know that the surrounding landscape influences the structure and functioning of rivers and streams, and that broad-scale catchment characteristics affect local-scale hydrology, habitat and biota (Frissell et al., 1986). The River Continuum Concept (Vannote et al., 1980) and Serial Discontinuity Concept (Ward and Stanford, 1983) highlight the importance of upstream-downstream linkages in lotic systems. These concepts were extended to studies on riverine-floodplain and riverine-terrestrial linkages that demonstrated the importance of maintaining lateral connections and dynamics between floodplain, terrestrial interfaces and river channels (Junk et al., 1989; Naiman and Décamps, 1990). The Hyporheic Corridor Concept emphasizes the role of surface-subsurface water linkages (Stanford and Ward, 1993). The central role of the natural flow regime in the dynamic structure and functioning of lotic systems is also widely recognized (Poff et al., 1997). Studies in fish conservation highlight the need for maintaining the temporal variation of natural flow regimes as well as impacts across a range of spatial scales from the micro-habitat to the entire catchment (Moyle and Yoshiama, 1994; Fausch et al., 2002).

These concepts build a good foundation for conserving freshwater biodiversity, emphasizing the need to consider processes and impacts across multiple spatial and temporal scales. They have been assimilated into conventional approaches to water ecosystem management such as river health assessment and monitoring, which is aimed at managing ecological integrity. Indeed, many calls for conserving freshwater biodiversity were fuelled by a surge of research on the ecological integrity of river ecosystems in the 1980s and 1990s, which provided quantitative evidence of the high levels of degradation (Norris et al., 2007). A large body of research in assessing and monitoring the ecological integrity of freshwater ecosystems now exists, forming the foundation of many national bioassessment programmes, such as the River InVertebrate

Prediction And Classification System (RIVPACS) in the United Kingdom (Wright et al., 1993); Environmental Monitoring and Assessment Program (EMAP) for surface waters in the United States (Lazorchak et al., 2000); AUStralian RIVER Assessment Scheme (AUSRIVAS) in Australia (Davies, 2000); and the River Health Programme (RHP) in South Africa (e.g. River Health Programme, 2006). These programmes have been running for several years now, and the next logical progression from monitoring the ongoing degradation of freshwater ecosystems is implementing actions to stem this problem. While bioassessment data can provide valuable insight, there is a need to assimilate these data into a strategic and systematic planning framework aimed directly at addressing and reversing the downward decline of freshwater biodiversity.

Integrated water resources management is a widely-acknowledged framework within which this need could be addressed. It explicitly seeks to balance long term ecological, economic and social concerns in the way water resources are managed within catchments (Jones et al., 2003). In South Africa, this concept has been translated into progressive policy in the national Water Act (Act No. 36 of 1998) through (1) the creation of a “Reserve” for each river system which caters for basic human and ecological water needs prior to other water allocations; and (2) the devolution of authority to local catchment management agencies. Unfortunately, there is still a strong utilitarian focus in applying integrated water resources management – in South Africa and globally – and all too often biodiversity concerns are not explicitly or systematically incorporated into the resulting catchment management plans (Gilman et al., 2004). From a biodiversity viewpoint, strategic input is needed regarding how many freshwater ecosystems should be conserved, and which ones would ensure maximum conservation benefit at the lowest possible social and economic cost.

The above discussion highlights the need for a more explicit, systematic and strategic approach to conserving freshwater biodiversity. The growing demand for water resources for social and economic development means that choices will need to be made about which systems are most strategic for conservation action. This requires a strategic, coordinated and landscape-level planning approach to conservation, pursued within appropriate scientific, policy, and management contexts. A key point of departure is the systematic identification of those freshwater ecosystems of high conservation value. We

can draw guidance from over two decades of research and practice in systematic conservation planning that has been limited almost exclusively to terrestrial and, more recently, marine environments.

SYSTEMATIC CONSERVATION PLANNING

Systematic conservation planning originated within the context of locating formal protected areas, or reserves. It grew from the realization that the world's protected area systems are biased in the biodiversity they represent, most commonly favouring areas of low economic potential – such as those with relatively low human population, unproductive soils, steep slopes or high altitudes (Pressey, 1994). Early efforts in conservation planning were therefore focussed on devising methods to become more systematic in the way biodiversity is represented in protected area systems and more strategic in the way limited conservation resources are used (Kirkpatrick, 1983; Margules et al., 1988).

Briefly, these techniques involved setting quantitative conservation targets for representing mapped biodiversity features of a region (e.g. species localities or vegetation types) and then selecting a minimum number of sites required to achieve these targets, using the concept of complementarity (Kirkpatrick, 1983; Pressey, 1994). The concept of complementarity ensures that it is not just the site with the most features that is chosen, but rather the site that contains the most so far unrepresented features. Complementarity therefore helps to ensure efficiency in the number of selected candidate sites. The whole exercise of selecting minimum sets in this way can be automated through the use of complementarity-based conservation planning algorithms (Sarkar et al., 2006).

The strong focus on representation in protected areas – where biodiversity is represented, bounded and protected – had limited applicability to connected ecological units such as freshwater ecosystems (Dunn, 2003). Even in terrestrial settings, problems with this narrow approach to conservation are evident. First, ecosystems within protected areas are essentially “locked away” from human use. This has the consequence of polarizing the needs of people and conservation where in many instances, it may not even be necessary to restrict all human activities (Richter et al.,

2003). Second, protected areas alone are unlikely to adequately conserve the full variety of biodiversity. Rather, they should be regarded as cornerstones to biodiversity conservation that are supplemented with other conservation strategies (Margules and Pressey, 2000). Third, areas selected for representation often ignore natural processes that are essential for the long term persistence of biodiversity, e.g. large migration corridors across the landscape (Balmford et al., 1998).

In an effort to address these problems the scope of systematic conservation planning has expanded over the years. The focus on protected areas has been broadened to explicitly include a variety of conservation mechanisms that acknowledge the needs of both people and ecosystems, ranging from highly restrictive (e.g. protected areas) to less restrictive (e.g. conservation easements, land stewardship) mechanisms (Margules and Pressey, 2000). This means that, in addition to locating the most strategic protected areas, systematic conservation planning can now be used as a tool to inform land use decision making. The narrow view of planning for the representation of biodiversity has also been expanded to acknowledge the need to incorporate the natural processes that are vital to the persistence of biodiversity. Design criteria, such as connectivity, are thus explicitly recognized as a critical component of systematic conservation planning (Cowling et al., 1999).

As systematic conservation planning has advanced and expanded its scope, it has become conceptually more suitable to planning for freshwater biodiversity. Despite better applicability, uptake of systematic conservation planning principles by the freshwater realm has been slow. While broad principles may be applicable, the approaches to dealing with these principles need to be freshwater-specific (Dunn, 2003). The lack of freshwater-specific frameworks and tools for systematic conservation planning is one of the key factors hampering its application in the freshwater realm. There remains a need to develop such frameworks and tools, and test their application within the context of integrated water resources management.

AIMS OF THIS THESIS

This thesis aims to address the need to develop freshwater-specific frameworks and tools to enhance systematic conservation planning for freshwaters by:

1. Identifying the basic requirements for undertaking freshwater conservation planning (Chapter 2)

This chapter summarizes the overarching principles of systematic conservation planning. It then examines early progress in tackling these principles within a freshwater context, distilling some basic requirements for freshwater conservation planning, and suggesting a road ahead to address key challenges.

2. Developing systematic methods for assessing river ecosystem endangerment and protection levels at broad sub-continental scales (Chapter 3)

This chapter focuses on developing rapid, systematic assessment methods that are appropriate for freshwater ecosystems, and which also allow comparisons to be made of the state of biodiversity across terrestrial, freshwater and marine realms.

3. Devising systematic methods for addressing the freshwater biodiversity gaps prevalent in protected area systems (Chapter 4)

This chapter develops methods for addressing the gaps in formal protected area systems, aimed at expanding protected area systems in a way that is beneficial to both terrestrial and freshwater biodiversity.

4. Developing a framework for incorporating persistence into the design of freshwater conservation area networks (Chapter 5)

Acknowledging that protected areas can realistically only play a partial role in the overall conservation of freshwater biodiversity, this chapter turns its attention to designing a conservation area network within the context of integrated water resources management. Drawing from concepts developed in freshwater ecology and systematic conservation planning, a conceptual framework for planning for the persistence of freshwater biodiversity is developed.

5. Testing the application of this persistence framework to planning for both representation and persistence of freshwater biodiversity at a local scale relevant to integrated water resources management (Chapter 6)

This chapter explores how this persistence framework can be applied in the design of a multiple-use conservation area network that can be used to inform integrated catchment management.

Research and practice in conservation has tended to focus on terrestrial biodiversity; while water resources management has tended to have a more utilitarian focus. It is high time to elevate freshwater biodiversity concerns on the agendas of both these sectors. By developing common conservation frameworks around which the water and conservation sector can engage and debate, this thesis attempts to enhance the integration of freshwater biodiversity concerns into both these sectors. It begins by examining the applicability of existing conservation planning principles in a freshwater context, exploring early progress and highlighting challenges and key knowledge gaps that need to be addressed (Chapter 2). Attending to some of the gaps highlighted in this review, existing terrestrial frameworks and models are then adapted for use in freshwater settings at both national (Chapters 3 and 4) and local (Chapter 6) levels of planning.

The term “freshwater” ecosystem is used throughout this thesis, and refers to all inland water bodies whether fresh or saline, including rivers, lakes, wetlands, sub-surface waters and estuaries. However, this thesis focuses largely on river ecosystems, with wetlands, estuaries and groundwater receiving less attention. Expanding the scope of the frameworks developed here is a major research frontier.

The work presented in this thesis does not stand in isolation, and was undertaken in parallel to other research and development that has also focussed on building the appropriate policy, institutional and operational arrangements that are required to promote more effective conservation of freshwater biodiversity. While much of this work is ongoing, the work that has been completed, and in which I was involved, is listed in Box 1.

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CHAPTER 2.
PROGRESS AND CHALLENGES IN
FRESHWATER CONSERVATION
PLANNING

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PROGRESS AND CHALLENGES IN FRESHWATER CONSERVATION PLANNING

ABSTRACT

Freshwater ecosystems and their associated biota are among the most endangered in the world. This, combined with escalating human pressure on water resources, demands that urgent measures be taken to conserve freshwater ecosystems and the services they provide. Systematic conservation planning provides a strategic and scientifically defensible framework for doing this. Pioneered in the terrestrial realm, there has been some scepticism associated with the applicability of systematic approaches to freshwater conservation planning. Recent studies, however, indicate that it is possible to apply overarching systematic conservation planning goals to the freshwater realm although the specific methods for achieving these will differ, particularly in relation to the strong connectivity inherent to most freshwater systems. Progress has been made in establishing surrogates that depict freshwater biodiversity and ecological integrity, developing complementarity-based algorithms that incorporate directional connectivity, and designing of freshwater conservation area networks that take cognisance of both connectivity and implementation practicalities. Key research priorities include increased impetus on planning for non-riverine freshwater systems; evaluating the effectiveness of freshwater biodiversity surrogates; establishing scientifically defensible conservation targets; developing complementarity-based algorithms that simultaneously consider connectivity issues for both lentic and lotic water bodies; developing integrated conservation plans across freshwater, terrestrial and marine realms; incorporating uncertainty and dynamic threats into freshwater conservation planning; collection and collation of scale-appropriate primary data; and building an evidence base to support improved implementation of freshwater conservation plans.

KEY WORDS: River; systematic conservation planning; biodiversity; integrated water resources management

INTRODUCTION

Fresh water affects every activity and aspiration of human society and sustains all terrestrial and aquatic ecosystems (Millennium Assessment, 2005). Yet this valuable resource is in crisis (Dudgeon et al., 2006). Expanding populations and increased socio-economic development have led to the degradation of freshwater ecosystems worldwide, and there is growing evidence that freshwater biodiversity is now amongst the most endangered in the world (Jenkins, 2003). The current situation becomes even more alarming when future threats are considered. Globally, the societal demand for water is predicted to escalate exponentially (Zemen et al., 2006), and water supplies in many areas are threatened by global climate change (Malmqvist and Rundle, 2002).

The simultaneous loss of freshwater biodiversity and increase in human pressures on water resources demands that concerted actions be taken to halt the progressive deterioration of freshwater ecosystems and sustain the valuable services they provide. To be effective, these actions should be well-planned, aimed at managing water for both people and biodiversity, and supported by sound national and institutional policies and strategies. Systematic conservation planning offers a structured, efficient and scientifically defensible conservation framework for achieving these objectives through locating priority geographic areas and implementing appropriate conservation actions (Margules and Pressey, 2000). It recognises that, in a world of social and economic constraints, not all areas will be able to be protected and aims to stem the loss of biodiversity through prioritising areas for conservation action in a spatially efficient configuration. From a freshwater perspective, such an approach can provide guidance on how water resource management, landscape development and freshwater biodiversity conservation can be balanced within an integrated catchment management framework.

This type of systematic, strategic and landscape-level planning for freshwater conservation is currently lacking in most catchment management strategies (Gilman et al., 2004). Recent systematic conservation planning studies for freshwater ecosystems are starting to provide outputs to address this critical gap in catchment management. To support this important direction for conservation and management of freshwaters, this chapter introduces generic principles of systematic conservation planning, consolidates

concepts that may be applicable in freshwater settings, and considers freshwater-specific approaches that have recently been developed.

In this chapter, “biodiversity” is defined in its broadest sense, to include the variety of all ecosystems, biological assemblages, species and populations (Convention on Biological Diversity, <http://www.biodiv.org>). The term “conservation area” refers to any area selected to meet the goals of a conservation plan; these areas in the planning region together form a “conservation area network” to promote the representation and persistence of biodiversity. The term “conservation area” should not be confused with “protected area”, which represents one option on a management continuum, ranging from strong restrictive use (the usual case for protected areas) to more open access in which only certain types of activities are managed (*sensu* Abell et al., 2007).

SYSTEMATIC CONSERVATION PLANNING APPROACHES

Systematic conservation planning emerged in the 1980s with the aim of conserving biodiversity over the long term, and several planning frameworks have since been developed (Margules and Pressey, 2000; Groves, 2003; Margules and Sarkar, 2007; Moilanen, 2008). All systematic approaches to conservation planning share three overarching principles - representation, persistence and quantitative conservation target setting. Representation refers to the need to adequately conserve the full variety of biodiversity features in a planning region, whilst persistence requires maintenance of the natural processes that support and generate biodiversity. Setting quantitative conservation targets is a defining characteristic of systematic conservation planning and can include, for example, the number of occurrences of a particular river type, the number of hectares of a specific wetland type, or the number of occurrences of a species. Conservation targets promote the design of spatially efficient conservation areas by providing a quantitative means for evaluating complementarity of candidate sites. This concept of complementarity – where conservation areas are chosen to complement each other in their biodiversity content – forms the computational backbone of most systematic conservation planning tools (Sarkar et al., 2006). Complementarity of a site is calculated as the contribution it makes to conservation targets not yet achieved in the existing set of conservation areas. This value is a relative

measure that needs to be recalculated each time a new site is added to the conservation area network.

These three principles – representation, persistence and conservation targets – set systematic conservation planning approaches apart from earlier scoring approaches to conservation prioritization that assess individual sites according to several biodiversity and management criteria (Dunn, 2003). While some scoring approaches take persistence criteria explicitly into account (Moilanen, 2008), most scoring approaches undermine representation (when ecosystem types with a naturally low score are not represented) and frequently result in spatial inefficiencies due to the lack of explicit consideration for complementarity.

Increasingly, a fourth principle for promoting effective and sustained implementation has been explicitly included in systematic conservation planning frameworks (Knight et al., 2006a). This principle addresses management actions in priority areas, and also confronts non-spatial issues that influence sustained and effective implementation. These issues include developing mechanisms for cross-sectoral cooperation, building capacity in conservation agencies, raising awareness of the need for conservation, and developing an appropriate monitoring and evaluation system.

Pioneered in the terrestrial realm, there has been some scepticism associated with the applicability of systematic approaches to freshwater conservation planning (Dunn, 2003). More recently, however, systematic conservation planning concepts have been applied in the development of freshwater-specific conservation planning frameworks (Abell et al., 2002; Higgins, 2003; Fitzsimons and Robertson, 2005; Roux et al., 2006). These existing frameworks have been used here to distil six basic requirements for freshwater conservation planning (Table 2.1) that are used to focus the rest of the discussion.

Table 2.1 Six basic requirements for freshwater conservation planning. Drawn from Abell et al. (2002), Higgins (2003), Fitzsimons and Robertson (2005), and Roux et al. (2006). These tasks do not provide a comprehensive conservation planning framework, but serve to focus the discussion around freshwater-specific approaches to tackling the most basic requirements in conservation planning. For more comprehensive generic frameworks to systematic conservation planning the reader is referred to Groves (2003) and Margules and Sarkar (2007).

Plan for effective and sustained implementation

- *Identify and involve key stakeholders*
- *Assess social, economic and institutional contexts*
- *Promote cooperation across all political boundaries and sectoral interests*
- *Develop a shared long-term regional vision and strategy at the catchment scale*

Evaluate current impacts and future threats

- *Use site-based ecological integrity data where available*
- *Use data on existing water use*
- *Supplement with remotely-sensed and mapped land cover data*

Plan for representation of freshwater biodiversity

- *Delineate freshwater systems and their associated catchments/sub-catchments*
- *Map biodiversity surrogates (e.g. species data, modelled species distributions or ecosystem types)*

Set quantitative conservation targets

- *For species and/or ecosystem types*

Plan for persistence

- *Incorporate connectivity*

Design a conservation area network

- *Design for spatial efficiency*
 - *Design for cost efficiency*
 - *Interpret within the context of multiple use zones*
 - *Integrate terrestrial and freshwater conservation plans*
-

PLANNING FOR EFFECTIVE AND SUSTAINED IMPLEMENTATION

The value and impact of a freshwater conservation plan can only be realized through its effective implementation. Responding from the outset of the planning exercise to key issues that enable implementation can greatly assist this process (Knight et al., 2006a). The lack of evidence-based studies on how best to support the effective implementation of freshwater conservation plans is a challenge in this young and emerging field. However, there are a number of lessons from both integrated water resources management (WWF, 2003) and terrestrial conservation planning (Knight et al., 2006b) that should guide planning for implementation in the context of freshwater conservation, including: (1) Identifying and involving key stakeholders in the planning process. This step is aimed at enhancing the sense of ownership of the plan. Stakeholders usually include implementing agencies and key interest groups who will be affected by the planning outcomes. (2) Assessing social, economic and institutional contexts to inform subsequent planning approaches and strategies. Such an assessment should provide contextual insight into the societal values, institutional capabilities, legislative framework and governance models, overlapping mandates, and degree of cooperation that exist in the domain (Pahl-Wostl, 2007). (3) Promoting cooperation across all political boundaries and sectoral interests (e.g. agriculture, urban development, mining, navigation, fisheries management and conservation). Designing and facilitating a social process whereby parties can learn together to develop a critical level of common understanding and intent are crucial for achieving coordinated action across sectors and levels of government (van Kerkhoff and Lebel, 2006). (4) Developing a shared long-term regional vision and strategy at the catchment scale to guide action at sub-catchment and local levels. This includes integration of policies, responsibilities, decisions, resources and costs.

Finding an appropriate lead agent to ultimately coordinate the implementation process is critical. The ideal lead agent would be an organization that is responsible for integrated catchment management. However, there are few examples globally of single organizations that are responsible for integrated catchment management. In reality, conserving freshwater ecosystems is usually dependent on cooperative efforts by

multiple agencies in the same catchment. Typically, lead agencies are responsible for biodiversity management or water resource management but seldom both. Nevertheless, a lead agency should have the capacity and credibility to coordinate management actions effectively, integrating and evaluating a variety of technical inputs from several disciplines, including freshwater ecology, conservation biology, socio-economics, hydrology, water quality and engineering. In addition, the capacity and credibility of a lead agent will be significantly strengthened over time if implementation of the plan is explicitly linked to a structured and iterative process of learning and decision making, such as advocated by adaptive management (Folke, et al. 2005).

EVALUATING CURRENT IMPACTS AND FUTURE THREATS

Assessing impacts and threats directly informs conservation strategies, management options and priorities for actions (Linke et al., 2007). Where possible, impacts and threats to both ecosystem and population persistence should be evaluated. However, data to inform assessments of population viability are seldom available. Freshwater conservation plans therefore rely mainly on assessing ecological integrity of ecosystems, based on the notion that ecosystems of high ecological integrity support and maintain the full natural range of biological features and ecological processes (Karr and Chu, 1995). Hence, ecosystems of high ecological integrity should ideally be selected for conservation. However, the degradation of freshwater systems is so pervasive that this is not always possible; in these instances, the system with the best potential for restoration is usually selected (Higgins, 2003).

The concept of ecological integrity is well-established in freshwater ecology, and a large number of site-based physical, chemical and biological indicators have been established to measure ecological integrity (Boulton, 1999). Site-level data, however, are generally only available for major rivers; data are lacking for smaller streams, which are often the last refuges for much biodiversity (Chapter 3). Thus, mapping ecological integrity for conservation planning is largely dependent on the use of land cover data, existing data on land- and water-use, and expert knowledge. This should ideally be supplemented with available site-based data where possible.

Several studies have quantified land cover within catchments, sub-catchments and riparian buffer strips to infer information about factors that impact ecological integrity of freshwaters (Table 2.2). For example, extent and intensity of agriculture is used to infer information on water use, sedimentation, and chemical and nutrient pollution; dam and road density, and number of road-stream crossings are used to estimate the degree of hydrologic alteration and fragmentation; and data on the distribution of infrastructure and urban areas are used to infer information about water use and pollution (Abell et al., 2002). More direct information is often used when available, such as information on water quality and hydrological modification (Table 2.2).

Matteson and Angermeier (2007) provide a protocol to assess the ecological risk of human activities on river systems. They use the spatial distribution of current threats (based mainly on land cover), combined with an expert assessment of each of their impacts on river systems, and their intensity within a sub-catchment. While this approach focuses mainly on current impacts, it could also be extended to future threats. Future threats such as population growth, planned dams, resource extraction leases, water abstraction plans, estimated water demands and climate change are all important future risks to consider. In the absence of such data, land tenure and land capability have been used as surrogates to predict the vulnerability of sub-catchments to degradation of ecological integrity (Linke et al., 2007). These data were then applied in a strategic framework that included affording priority to sub-catchments with both a high conservation value and a high vulnerability to future threats.

PLANNING FOR REPRESENTATION

It is impossible to map and classify all the elements of biodiversity. Instead, conservation planners rely on surrogate measures of biodiversity (Rodrigues and Brooks, 2005) in the form of species, species groups or ecosystem types (Table 2.3). The coarse- and fine-filter approach incorporates this concept by including all ecosystem types (coarse-filter) as well as species that will not be well represented by ecosystems, such as those that are rare, endangered, occur locally, or are migratory (Groves, 2003). The coarse-filter premise holds that conserving representative

ecosystem units conserves many common species and communities, and the environments in which they evolve. Implementing this approach allows us to advance freshwater conservation beyond species as the only measure of biodiversity, to conserve habitats and ecosystems on a systematic basis.

Table 2.2 Surrogates that have been used in mapping landscape-level ecological integrity. Numbers in superscript refer to the following references: Abell et al. (2002)¹; Stein et al. (2002)²; Weitzell et al. (2003)³; Snyder et al. (2005)⁴; Linke et al. (2007)⁵; Matteson and Angermeier (2007)⁶; Norris et al. (2007)⁷; and Thieme et al. (2007)⁸.

Remotely-sensed and mapped land cover data

Natural vegetation^{1,3,4,5,7,8}
 Railway and road density^{1,2,3,5,6,7,8}
 Railway- and road-stream crossings^{1,3,6,7}
 Impervious surface^{3,4}
 Impoundments^{1,2,3,6,7}
 Mines and quarries^{1,2,3,6,8}
 Urban and industrialised areas^{1,2,5,6,8}
 Plantation forestry/logging^{1,2,5,8}
 Irrigated agriculture^{1,2,4,5,6}
 Grazing agriculture^{1,2,5,6}

Other available data

Bioassessment data⁷
 Conservation areas⁵
 Point source pollution (e.g. waste sites)^{2,3,6}
 Dam capacity²
 Population density^{2,3}
 Flow diversions²
 Distribution of barriers such as levees and weirs^{1,2,7}
 Hydrological alteration⁷
 Suspended sediment and nutrient loads^{5,7}
 Acidification⁵
 Salinity⁵
 Extent of alien plant infestation⁵

Table 2.3 Examples of biodiversity surrogates and quantitative conservation targets used in recent freshwater conservation plans.

Type of biodiversity surrogate	Method of setting target and references
<i>Species</i>	
Fish species	At least two spatially distinct occurrences of each target species within each Ecologically Distinct Unit (Sowa et al., 2007); or at least one “viable” population of all focal fish species in the planning domain, based on point locality distributions (Roux et al. 2008); or at least two viable* populations of all indigenous fish species in the planning domain, preferably in different sub-catchments (Chapter 6).
Invertebrate groups	At least one occurrence of each benthic macroinvertebrate group (identified to species level where possible), based on modelled distributions and probability of occurrence (Linke et al., 2007); or at least one occurrence of an invertebrate genus, based on point locality distributions (Roux et al., 2008).
Aquatic water beetle species	At least one occurrence of each species, taking viability into consideration by using only areas where ≥ 2 capture records have been documented (Abellán et al., 2007).
<i>Habitat types</i>	
River types	20 % of the total length of each river type in the planning region, expressed in km (Chapter 6; Roux et al., 2008); or, 20 % of the total area of each stream habitat type in the planning region contained within a GIS buffer of 10 km either side of the river, expressed in km ² (Thieme et al., 2007); or at least one occurrence of each stream habitat type (Weitzell et al., 2003).
Wetland types	20-25 % of the total area of wetland type within the planning domain, expressed in km ² (Thieme et al., 2007).
Sub-catchment habitat types	20% of the total area of sub-catchment habitat types within the planning region, expressed in km ² (Thieme et al., 2007).
<i>Processes</i>	
Upstream connectivity	Non-headwater sub-catchments cannot be protected without their upstream sub-catchments, automated into decision-support design rules (Linke et al., 2007).
Fish migratory routes	100 % of all migratory routes between earmarked fish sanctuaries required for fish species target achievement, added in manually by regional fish specialists once sanctuaries had been identified (Chapter 6).
Free-flowing rivers	Maintain free-flowing rivers along two major rivers in the planning region (Thieme et al., 2007); or 100 % of all un-dammed major rivers within the planning region (Chapter 6).
Significant water yield areas	The 20% of sub-catchments that generate of the highest mean annual run-off in the planning region, expressed as number of sub-catchments (Driver et al., 2005)

*Here, viable was judged using expert opinion and was interpreted as a reproducing and self-maintaining population

Incorporating species

Use of species surrogates is dependent on relatively comprehensive species inventories, and even these are fraught with omission errors that fail to detect species where they truly occur (false absences). Omission errors can be addressed through the use of models that compile continuous geographic distributions for selected species based on their environmental relationships. For example, generalised additive models that cope with nonlinear relationships between species and environmental predictors have been used to map the distributions of several benthic macroinvertebrates (Linke et al., 2007), and both linear and nonlinear models have been used to map the distribution of freshwater fish (Filipe et al., 2004; Leathwick et al., 2005; Sowa et al., 2007). Although these modelling techniques address the problem of omission errors, they fall prone to commission errors (false presences) – areas where species are modelled as present when they are actually absent. In conservation planning, commission errors are more serious, as they may lead to considering a species conserved when, in reality, it is not. Applying stringent thresholds to the probabilities of occurrence generated from modelled species distributions helps to avoid such errors (Wilson et al., 2005). This should ultimately be followed by on-ground inspection of all sites selected for conservation, to verify that the features for which the site was selected are indeed present.

Generating species models, one species at a time, can be very time-consuming for conservation planning over large regions. The recent application of multivariate adaptive regression splines (MARS) appears to be able to circumvent this challenge (Leathwick et al., 2005), fitting a single model to multiple species. Using 15 freshwater fish species, an evaluation of model performance for multi-species analysis indicated a comparable performance to models fitted individually (Leathwick et al., 2006). This technique also enables statistical mapping of ecosystems, informed by multi-taxon biological survey data.

Incorporating ecosystems

Mapping and classifying freshwater ecosystem types requires first delineating actual river networks and non-riverine ecosystem types, such as lakes, wetlands and estuaries.

The availability of high-resolution digital elevation data for the world (Shuttle Radar Topography Mission data; see <http://srtm.usgs.gov/>) and GIS hydrological modelling tools, such as ArcHydro (Maidment, 2002) and HydroSHEDS (Lehner et al., 2006), have been used to delineate catchments, sub-catchments and stream networks (Thieme et al., 2007). In contrast, the delineation of wetlands and estuaries across large planning regions has been more problematic. Traditionally, wetland inventories have been based on local knowledge and interpretation of aerial photographs; this is time-consuming when conducted for large regions. Imagery from airborne and satellite remote sensing platforms that are used in conjunction with spatial modelling technologies that enhance wetland detection show promise in addressing this problem (Kingsford et al., 2004; Ausseil et al., 2007; Davidson and Finlayson, 2007; Hamilton et al., 2007), with the former two studies reporting a 70-86 % overall accuracy in detecting wetlands and classifying them according to three broad geomorphic groups (palustrine and riverine, estuarine, and lacustrine).

Once freshwater features have been delineated, they can be classified into ecosystem types. Tests of surrogates in the terrestrial realm (see Rodrigues and Brookes, 2007 for review) indicate that, data permitting, it is preferable to derive ecosystem classifications that are informed by biological data (e.g. species distributions, vegetation types, ecoregions), rather than to base them solely on abiotic environmental data (e.g. geology, climate). Snelder et al. (2007) provide an example for developing a river environmental classification for New Zealand which is statistically informed by fish species survey data. Extending this protocol to include a wider range of taxa (e.g. invertebrates and plants) holds potential.

Most other freshwater conservation plans that have mapped ecosystem types use a combination of broad-scale biological assemblage data (e.g. vegetation or freshwater ecoregions) and abiotic data. At a landscape level, abiotic data such as geology, climate, topography and vegetation can be used to infer hydrological and geomorphological characteristics such as flow variability, channel morphology, substratum and broad water quality (Figure 2.1). Stream reaches can be classified to a finer level of detail (Figure 2.1) according to similarities in flow and sediment transport regimes, for

example, by using flow variability measures, stream size, channel gradient and valley dimensions (Dollar et al., 2007).

Choice of surrogates

The choice of surrogates will ultimately depend on the data available. However, lessons emerging from testing the effectiveness of surrogates in terrestrial conservation planning indicate that there is no single effective surrogate and that plans based on multiple surrogates are more effective at capturing the full variety of biodiversity (Rodrigues and Brooks, 2007). Figure 2.1 shows an example of the data that can be used to derive ecosystem types in data poor regions. In data rich planning regions, this approach can be extended to incorporate the coarse-fine filter approach to surrogates. For example, Sowa et al. (2007) derived 158 freshwater conservation opportunity areas for the state of Missouri, USA. Here, river types included not only physical data, but information on biological assemblages, for example unique zoogeographic zones and distinct ecological drainage basins such as those described in Higgins et al. (2005). In addition, these river types were supplemented with modelled distributions of 32 crayfish, 67 mussel, and 216 fish species. A similar coarse-fine filter approach has been applied in a data rich area of South Africa (Roux et al., 2008), where river types were supplemented with actual fish species occurrences rather than modelled distributions.

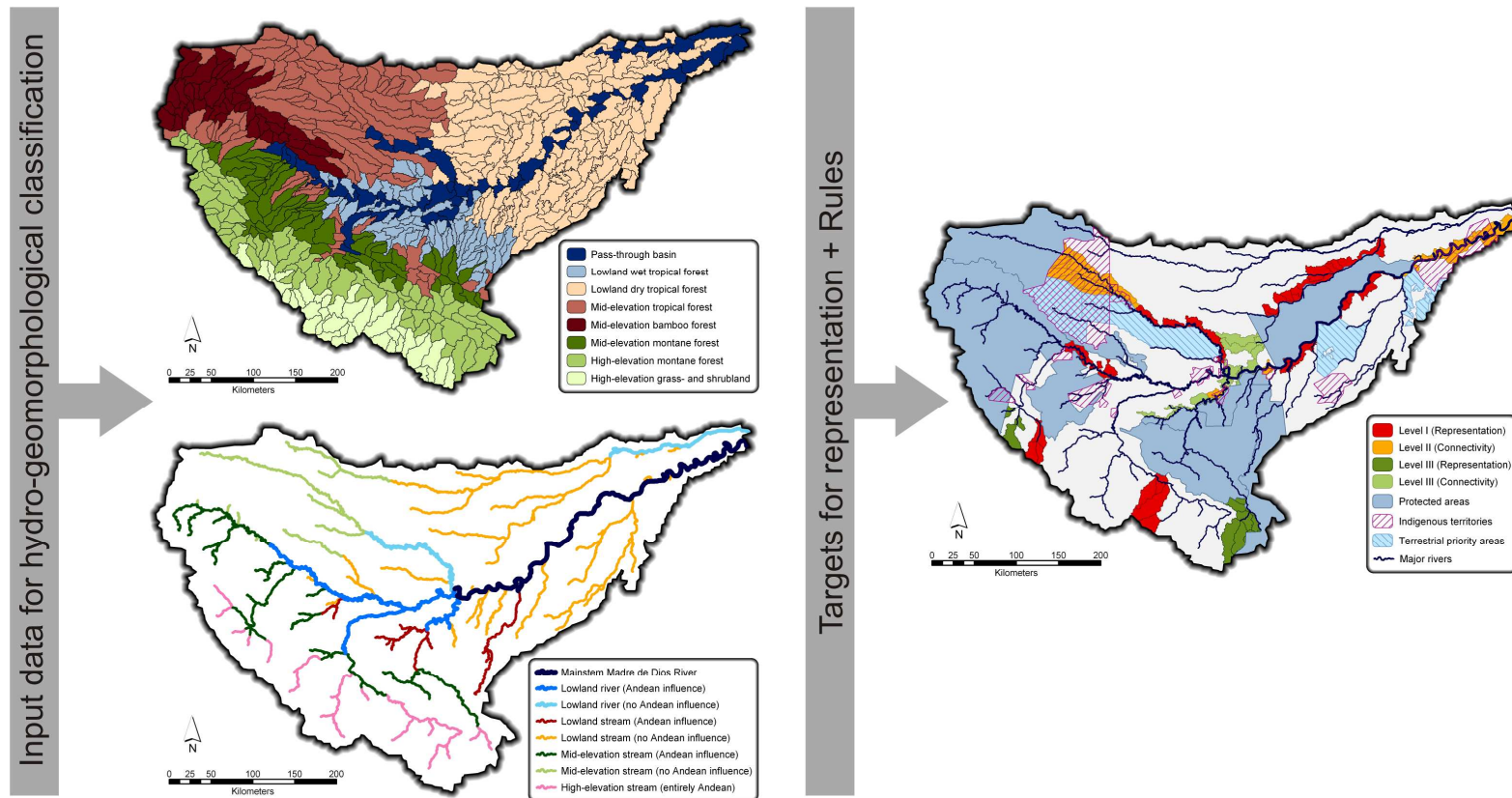


Figure 2.1 An example of a freshwater conservation plan for a data poor region in a remote Amazonian catchment, after Thieme et al. (2007). Physically-defined river and floodplain types were derived using a hierarchy of available data that describe hydro-geomorphological characteristics of streams (e.g. elevation, modelled surface runoff, geology). For each of these 22 habitat types, targets were set at 20 % of the total extent (see Table 2.3). In meeting targets, choices were guided by objectives to maximize complementarity and connectivity, choose the most intact systems, and align with terrestrial conservation priority areas and existing protected areas. The final integrated conservation plan differentiates between areas that may require different management strategies: Level I areas are relatively intact and a range of protection mechanisms can be employed; Level II areas coincide with indigenous territories where conservation will depend on collaboration with indigenous groups; and Level III areas experience high use and thus require active threat mitigation to meet conservation needs.

SETTING QUANTITATIVE CONSERVATION TARGETS

Setting scientifically defensible conservation targets is challenging because minimum population sizes or minimum habitat requirements for most freshwater species are not known. In the absence of empirical data, several different methods have been used in the freshwater realm (Table 2.3). Species targets are generally based on absolute occurrences, using either point locality data (Weitzell et al., 2003; Roux et al. 2008), or predictive models and probability of occurrence (Linke et al., 2007; Sowa et al., 2007). Species targets that incorporate multiple occurrences in different catchments (Chapter 6; Sowa et al., 2007) provide opportunity for different genetic lineages to be conserved.

Habitat type targets (e.g. amount of each river, wetland or estuary type) are frequently based on a recommendation made by the Caring for the Earth Strategy (IUCN, 1991) of maintaining, and restoring where necessary, at least 20 % of each habitat type. Such proportional targets should be based on pre-settlement extents of each habitat type rather than current extents (Pressey et al., 2003). The manner in which habitat targets are calculated varies among studies (see Table 2.3), and has been expressed in terms of length, area or number of occurrences. The 20 % target is an arbitrary, over-simplified and uniform measure that does not take into account the specific requirements of different ecosystem types. It should be applied with caution, as evidence-based studies on thresholds suggest requirements that are in many instances nearly three times higher (Svancara et al., 2005).

Nevertheless, a quantitative target that has been officially endorsed and effectively communicated has a powerful ability to inform and direct policies, processes, programmes and actions. Thus, preliminary targets should be incorporated into planning processes as early as possible, but with the recognition that these are hypotheses that should be tested and refined as better empirical data and methods for target setting become available (Pressey et al., 2003).

PLANNING FOR PERSISTENCE

Planning for persistence refers to the incorporation of natural processes into conservation plans to ensure that biodiversity persists and evolves naturally over time. Nearly all patterns and processes in freshwater ecosystems are underpinned by connectivity along three spatial dimensions (longitudinal, lateral and vertical), and a temporal dimension (Pringle, 2001). This strong connectivity gives rise to some of the most profound challenges in conserving freshwater ecosystems. The fluid and fugitive nature of freshwater ecosystems, combined with their position in the landscape (generally the lowest point), mean that they are “receivers” of upstream, downstream and upland impacts (Dudgeon et al., 2006), and require the adoption of a whole-catchment management approach.

Incorporating longitudinal connectivity

Longitudinal connectivity refers to the upstream-downstream continuum of lotic systems, where conservation of their lower reaches is largely dependent on the conservation of upstream reaches and vice versa. There are two categories of challenge associated with incorporating longitudinal connectivity into freshwater conservation plans.

The first pertains to the difficulty of implementation when dealing with large rivers or species that migrate over long distances, as this requires management of vast landscapes. Moreover large rivers tend to be heavily dammed to provide water and hydro-electric power (Nilsson et al., 2005) and cannot be “locked” away from human use. Whether conserving a pristine or utilized river, conservation action over large catchment areas will require the coordinated action of multiple stakeholders. In many instances, conserving a pristine large river may be out of the question, and maintaining longitudinal connectivity will require attention to providing fish passage to barriers such as dams and weirs (Schilt, 2007), and developing a beneficial environmental flow regime (Tharme, 2003).

The second challenge relates to the technical aspects of designing spatially efficient conservation area networks. Incorporating longitudinal connectivity has

commonly been accomplished after selecting areas required for representation, through incorporating their upstream and downstream sub-catchments. While it is relatively simple to establish connectivity areas in this way, it seldom yields the most spatially efficient solution because the complementarity between representation and connectivity areas is not explicitly quantified. This problem has recently been addressed through the use of complementarity-based conservation planning algorithms in which rules are set for achieving longitudinal connectivity whilst simultaneously achieving representation (Linke et al., 2007; Moilanen et al., 2008). These are discussed in further detail in the section below on designing for spatial efficiency.

Incorporating lateral connectivity

Lateral connectivity refers to the interconnection of ecological zones from the river channel to its riparian zone and associated wetlands, and the surrounding catchment (Naiman and Décamps, 1997). In a spatial design, lateral connectivity is incorporated in its broadest sense by using sub-catchments as planning units (Higgins et al., 2005; Thieme et al., 2007). Selecting a freshwater feature, such as a river reach or wetland, consequently selects the entire sub-catchment within which the feature falls, highlighting the need to manage the freshwater feature, as well as the surrounding land and network of smaller streams.

At finer scales of freshwater conservation planning, varying levels of lateral connectivity within a sub-catchment can also be distinguished. For example, remote sensing techniques have been applied to distinguish riparian zones (Goetz, 2006), and wetlands and floodplains (Hamilton et al., 2007). Although not ideal, in the absence of such spatial delineations, conservation plans can cater simplistically for varying levels of lateral connectivity within a sub-catchment by using buffer zones surrounding the freshwater ecosystems of concern (Saunders et al., 2002).

Incorporating vertical connectivity

The persistence of many freshwater ecosystems and their associated biota is maintained by their hydrological linkages with groundwater (Sophocleous, 2002). The

incorporation of vertical connectivity into conservation plans requires, at a minimum, the scale-appropriate mapping of spatial patterns of groundwater discharge and recharge. This enables identification of areas where activities that reduce groundwater quantity or quality need to be managed, e.g. by controlling or preventing groundwater abstraction, maintaining natural vegetation cover, and clearing water-consuming alien invasive plants.

A modelling approach that predicts areas of high groundwater-surface water interaction has recently been applied to a semi-arid area of South Africa (Conrad and Münch, 2006) where groundwater plays an important role in sustaining river base flows and supporting refuge pools in the dry season. These areas were incorporated into a regional conservation plan (Chapter 6) to highlight sub-catchments where groundwater resources should be strictly managed. Similarly, Baker et al. (2003) have developed a landscape-based modelling approach to map groundwater dynamics and identify areas of groundwater recharge and discharge.

DESIGNING A CONSERVATION AREA NETWORK

Multiple criteria inform the design of a conservation area network, including the representation of all biodiversity features, at least to the level of their minimum conservation targets; incorporating important design issues that promote the persistence of targeted biodiversity; the consideration of socio-economic opportunities and constraints in the region; and alignment with other conservation and planning initiatives. A conservation area network should build on the legacy of existing conservation initiatives in the region (e.g. protected areas) and be as economical as possible, considering both spatial and implementation costs.

Designing for spatial efficiency

Spatial efficiency is achieved through the concept of complementarity. For example, where there is a choice of several areas for achieving representation of a specific biodiversity feature, an area that contributes to as many other under-represented

biodiversity features as possible should be chosen. Alternatively, an area that is going to be necessary for achieving upstream-downstream connectivity should be chosen.

Many existing freshwater conservation plans were designed in a series of manual steps, taking cognisance of complementarity in design, and opportunities and constraints within the region (e.g. Weitzell et al., 2003; Sowa et al., 2007; Thieme et al., 2007 – see Figure 2.1). There are also some examples that have applied generic complementarity-based conservation planning tools to aid decision making on the most efficient spatial design (Nel et al., 2006). However, these generic tools have been developed in the terrestrial realm and their application is limited for freshwaters by their inability to adequately address issues of directional connectivity. Addressing this problem, Linke et al. (2007) have recently adapted a step-wise heuristic complementarity algorithm that has traditionally been used in terrestrial conservation planning by setting a rule that all sub-catchments upstream of selected non-headwater sub-catchments must be included in the selection.

This represents an important advance toward the development of automated freshwater conservation planning tools. However, a potential problem with the use of step-wise heuristic algorithms is that they are implemented sequentially: once a site has been selected it is locked into the final configuration, regardless of whether a more efficient configuration could be attained when connectivity for later selections is considered. If a conservation design is locked into a specific configuration too early in the planning process, it can lead to inefficiency, especially when strong connectivity rules are applied. Two approaches used to address this problem are rapidly gaining ground in the development of conservation planning tools: (1) algorithms that focus on full optimization, finding the best solution by comparing every other possible solution (Rodrigues and Gaston, 2002); and (2) metaheuristic algorithms that implement random substitution of sites in the beginning of the selection process, to test if more efficient spatial configurations across the planning domain can be achieved (Sarkar et al., 2006). In a study to optimize the removal of fish barriers in rivers, O'Hanely and Tomberlin (2005) demonstrate how both these approaches can be applied to rivers. Perhaps one of the most significant advances in this field is the recent adaptation of a generic conservation planning algorithm to include directional connectivity (Moilanen et al.

2008). These advances bode well for the development of generic conservation planning tools for freshwaters.

Designing for cost efficiency

Given the increasingly fierce competition for water resources, freshwater conservation plans should ideally incorporate an explicit assessment of conservation costs. Freshwater conservation priorities determined on the basis of ecological, social and economic costs enable explicit examination of the trade-offs between conservation and utilization, align more closely with the goals of integrated water resource management, and are more likely to secure commitment from politicians and decision-makers (Dudgeon, et al. 2006). Naidoo et al. (2006) highlight several conservation costs to consider when selecting conservation areas, including acquisition costs, management costs, transaction costs (e.g. from negotiating with individual land owners) and opportunity costs.

For freshwaters, the process of examining conservation costs is complex as it involves understanding the costs of conserving a particular area, as well as the costs associated with the water required to sustain that area over time. This involves quantifying the accrual of benefits and costs across entire catchments. A large variety of formal methodologies have been developed within the field of environmental flow assessment that evaluate alternative water allocations within a catchment (Tharme, 2003). Essentially, these methodologies assess trade-offs in flow requirements that would sustain both human and biodiversity needs for water resource management. The integration of such methodologies with conservation planning holds immense potential, but to date this has not been accomplished. Several multiple criteria decision-making methods have also been used in terrestrial conservation planning to aid identification of a preferred scenario based on qualitative or quantitative ecological, social and economic costs (e.g. Moffett et al., 2006). The applicability of these to freshwater conservation planning and integrated water resource management should be assessed.

Including multiple land-use options

Incorporating connectivity often creates space-hungry conservation plans, whose cost and complexity may overwhelm implementation agencies and politicians, and thus hamper their implementation. While the challenges of implementing conservation actions over large, multi-stakeholder areas remain immense, spatial planning offers an opportunity to ease this challenge through multiple-use zoning. This helps to make a conservation plan politically more “palatable”, as use restrictions can be adapted to the specific function of the zone and do not necessarily exclude all uses. For example, sub-catchments selected for representation can be afforded a high level of protection, where uses are fairly restrictive; whilst areas required for connectivity may be able to withstand some level of utilization (Chapter 6; Thieme et al., 2007).

Even within sub-catchments, different zones can be distinguished. For example, riparian zones of selected catchments can be allocated a higher level of protection than the sub-catchment as a whole (Saunders et al., 2002). A recent hierarchical protection strategy for freshwaters has incorporated this notion, embedding “Freshwater Focal Areas” and “Critical Management Zones” within “Catchment Management Zones” (Abell et al., 2007). Freshwater Focal Areas describe the location of a specific freshwater feature requiring protection, where management is likely to be fairly restrictive to prevent direct disturbances to the feature. Critical Management Zones describe those areas where management is essential for maintaining functionality of a focal area, and restrictions are likely to depend on the function of that zone. A Catchment Management Zone describes the entire upstream catchment of a critical management zone. Applying this concept (Figure 2.2), zones within each sub-catchment selected for conservation can be further distinguished such that systems selected for representation of a particular ecosystem type or species are the Freshwater Focal Areas; the Critical Management Zones are a mixture of riparian zones for maintaining connectivity, groundwater management zones, and wetland buffer zones; and the entire sub-catchment is managed as a Catchment Management Zone.

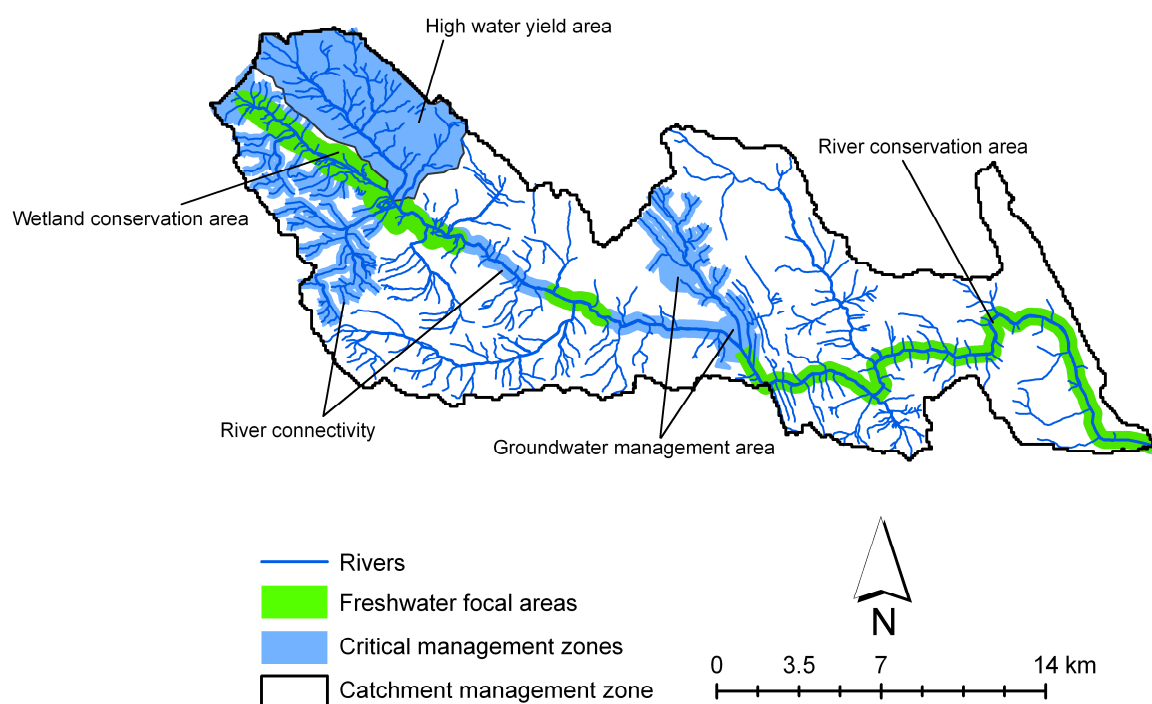


Figure 2.2 Schematic of potential freshwater conservation zones, after Abell et al. (2007). “Freshwater Focal Areas” required for representation are embedded within “Critical Management Zones” that support these focal areas, which are in turn nested within “Catchment Management Zones” that describe the entire upstream catchment of a Critical Management Zone.

Integrating terrestrial and freshwater conservation plans

This chapter has taken a deliberately one-sided view of biodiversity in an attempt to explore the explicit incorporation of freshwater biodiversity into conservation planning. Embracing a more holistic view of conservation planning in which freshwater, land and sea are managed in an integrated manner will ultimately be most effective at conserving the full spectrum of biodiversity on Earth. This will require promoting integrated policy and management strategies as well as designing more integrated spatial outputs.

Designing integrated conservation area networks may require more than simply adding sets of selected priority areas together (Abell, 2002), particularly if complementarity is to be considered. An integrated conservation plan will also need to derive useful integrated products for management authorities without losing the nuances

specific to each realm. A planning exercise in the Upper Mississippi River Basin has dealt with this issue by developing two separate products: (1) a detailed freshwater conservation plan to serve as a source of information for comprehensive management action; and (2) a combined map of all terrestrial and freshwater priorities for directing coordinated conservation action throughout the region (Weitzell et al., 2003). Attempts have also been made to align freshwater conservation plans with terrestrial priorities by preferentially selecting areas adjacent to existing protected areas or terrestrial priority areas (Figure 2.1).

Different scenarios may be applicable in different situations. For example, in planning regions where very few options remain for conserving intact freshwater ecosystems and their associated biota, integrated planning efforts that use freshwater features as their foundation and are subsequently expanded to achieve residual targets for terrestrial features may be a more effective and efficient way of designing conservation area networks. On the other hand, in planning regions where several freshwater options remain, achieving terrestrial and freshwater conservation targets simultaneously may result in better complementarity. In areas where marine ecosystems are strongly affected by land and freshwater linkages, it may be feasible that priority marine ecosystems drive initial selections.

CONCLUSIONS AND FUTURE RESEARCH

Systematic planning for freshwater conservation has finally started gaining momentum. This is long overdue given the large-scale degradation of freshwater ecosystems, the projected increase in human pressures on these ecosystems and the immense importance of freshwater ecosystems in providing sustainable ecosystem service delivery. Recent studies suggest that it is possible to apply overarching systematic conservation planning principles to the freshwater realm, although methods will be freshwater-specific particularly in relation to connectivity. Progress has been made in the development of surrogates that depict freshwater biodiversity and ecological integrity, and there have been some advances in the development of complementarity-based algorithms that incorporate directional connectivity. However, considerable effort is still required to

provide explicit frameworks for freshwater conservation planning, and to improve both the implementation of freshwater conservation plans and their scientific rigour.

Improving implementation will be an on-going, long-term investment of developing evidence-based case studies, monitoring and evaluation, and feedbacks into adaptive management. Scientists will need to become closely involved in the social process of implementation, supporting the spirit of co-learning and adaptive management. Evidence of the benefits of conserving freshwater ecosystems also needs to be made explicit (Gilman et al., 2004) and communicated to decision-makers (Dudgeon et al., 2006).

Research to improve the scientific rigour of freshwater conservation plans includes: (1) Strengthening the emphasis on planning for non-riverine freshwater systems, particularly for wetlands and groundwater. The incorporation of the latter into conservation planning is virtually non-existent. (2) Evaluating the effectiveness of freshwater biodiversity surrogates in conservation planning. Existing biodiversity surrogate evaluation protocols (Rodrigues and Brooks, 2007) should be evaluated for possible refinement and application. (3) Establishing scientifically defensible methods of setting conservation targets. Where freshwater species data exist, one approach would be to investigate species turnover along the length of a river system (*sensu* Desmet and Cowling, 2004), or species-discharge curves (*sensu* Xenopoulos and Lodge, 2006). (4) Developing generic complementarity-based algorithms that simultaneously consider connectivity issues for both lentic and lotic water bodies. Methods of linking these algorithms with socio-economic information that examine trade-offs between conservation costs and benefits and how these accumulate at a catchment-wide scale would be a significant advance. (5) Exploring options for designing resource efficient and integrated conservation area networks across terrestrial, freshwater and marine realms without losing realm-specific information. (6) Incorporating uncertainty and dynamic threats, such as climate change, into freshwater conservation planning. Early research (e.g. Xenopoulos et al., 2005; Palmer et al., 2008) suggests that building conservation areas that are resilient to climate change will require including management interventions such as drastically limiting water withdrawal in these catchments (particularly in regions prone to reduced discharge from climate change),

and removing barriers that inhibit altitudinal migration to optimal stream temperatures. (7) Finally, a concerted long-term research effort is also required for collection and collation of scale-appropriate, primary data on freshwater biodiversity and ecological integrity, particularly in data poor areas.

As with most emerging fields of applied science, a networked community of scientists and practitioners needs to be built to allow for the testing, exchanging and debating of various approaches, as well as the documentation and sharing of experiences, in line with the belief that we are more effective as a transdisciplinary group. Through collectively building on the approaches considered in this paper, and addressing some of the major research gaps and challenges outlined here, it is hoped that systematic conservation planning in the freshwater realm will evolve into an increasingly evidence-based and science-led process, one which leads to its widespread adoption and effective conservation of freshwater systems.

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CHAPTER 3.
RIVERS IN PERIL INSIDE AND OUTSIDE
PROTECTED AREAS: A SYSTEMATIC
APPROACH TO CONSERVATION
ASSESSMENT OF RIVER ECOSYSTEMS

University of Cape Town

RIVERS IN PERIL INSIDE AND OUTSIDE PROTECTED AREAS: A SYSTEMATIC APPROACH TO CONSERVATION ASSESSMENT OF RIVER ECOSYSTEMS

ABSTRACT

This study establishes a framework within which a rapid and pragmatic assessment of river ecosystems can be undertaken at a broad, sub-continental scale, highlighting some implications for achieving conservation of river biodiversity in water-limited countries. The status of river ecosystems associated with main rivers in South Africa was assessed based on the extent to which each ecosystem had been altered from its natural condition. This requires consistent data on river integrity for the entire country, which was only available for main rivers; tributaries were thus excluded from the analyses. The state of main river ecosystems in South Africa is dire: 84 % of the ecosystems are threatened, with a disturbing 54 % critically endangered, 18 % endangered, and 12 % vulnerable. Protection levels were measured as the proportion of conservation target achieved within protected areas, where the conservation target was set as 20 % of the total length of each river ecosystem. Sixteen of the 112 main river ecosystems are moderately to well represented within protected areas; the majority of the ecosystems have very low levels of representation, or are not represented at all within protected areas. Only 50 % of rivers within protected areas are intact, but this is a higher proportion compared to rivers outside (28 %), providing some of the first quantitative data on the positive role protected areas can play in conserving river ecosystems. This is also the first assessment of river ecosystems in South Africa to apply a similar approach to parallel assessments of terrestrial, marine and estuarine ecosystems, and it revealed that main river ecosystems are in a critical state, far worse than terrestrial ecosystems. Ecosystem status is likely to differ with the inclusion of tributaries, since options may well exist for conserving critically endangered ecosystems in intact tributaries, which are generally less regulated than main rivers. This study highlights the importance of healthy tributaries for achieving river conservation targets, and the need for managing main rivers as conduits across the landscape to support ecological processes that depend on connectivity. There is also a need for a paradigm shift in the way protected areas are designated, as well as the need for integrated river basin management plans to include explicit conservation visions, targets and strategies to ensure the conservation of freshwater ecosystems and the services they provide.

KEY WORDS: Conservation assessment, protected area gap analysis, conservation status, freshwater biodiversity, freshwater conservation planning, integrated river basin management

INTRODUCTION

Conserving river ecosystems depends on whole-catchment management, where land and water is managed in an integrated manner which aims to achieve ecological and socio-economic sustainability (O’Keeffe, 1989; Ward, 1998; Saunders, et al., 2002). This requires the development of integrative assessment and planning approaches that proactively consider the needs of both terrestrial and freshwater ecosystems. Systematic conservation assessment and planning methodologies are relatively well advanced for terrestrial ecosystems, both globally and in South Africa (Margules and Pressey, 2000; Cowling and Pressey 2003; Balmford, 2003; Driver et al., 2003; Groves, 2003). However, rivers have generally been poorly dealt with in most assessments of terrestrial ecosystems unless they are considered important for terrestrial biodiversity pattern and process, and their conservation status is usually ignored. In an effort to correct this, systematic conservation assessments and plans specifically targeting freshwater ecosystems have begun to emerge (e.g. Roux et al., 2002; Higgins, 2003; Weitzell et al., 2003), applying the basic concepts that have been developed for terrestrial ecosystems as well as recognising the need for some refinements to make the plans more suitable to the freshwater realm (Dunn, 2003). However, the majority of these assessments and plans are done in isolation to terrestrial ecosystem assessments and there is a need to combine these to develop assessments, plans, strategies and policies that are inclusive of both terrestrial and freshwater realms (Abell, 2002), to begin meeting the needs of integrated river basin management.

Thieme et al. (2005) have recently completed a continental scale assessment of freshwater ecoregions of Africa and Madagascar, which complements a terrestrial assessment of the same region (Burgess et al., 2004). Together, these shed light on a means of integrating assessments in that they both classify the respective freshwater and terrestrial ecoregions according to five levels of endangerment that are based on a similar logic as that used for threatened species in the IUCN Red Data Books (Mace and Lande, 1991; Hilton-Taylor, 2000). The advantage of using these endangerment categories for assessing both terrestrial and aquatic ecosystems are twofold: (i) they provide a familiar political terminology around which much species conservation policy has been developed (e.g. Mace and Lande, 1991; IUCN, 1994; IUCN, 2001), which

may therefore facilitate incorporation into existing policy mechanisms; and (ii) they provide a common currency for assessing ecosystems, thus enabling comparisons across terrestrial and aquatic realms, and the development of appropriate integrated strategies. Similar endangerment categories were used to assess freshwater ecosystems in this study, in an attempt to develop a common terminology for comparisons with assessments of terrestrial (Reyers et al., 2007), marine (Lombard et al., 2004) and estuarine (Turpie, 2004) ecosystems. An additional advantage of applying these endangerment categories in the context of this study is that South African biodiversity policy (National Environmental Management: Biodiversity Act No. 10 of 2004) provides for the listing of threatened ecosystems and this approach offers a means of identifying such ecosystems.

This study presents a nation-wide, sub-continental assessment of ecosystems associated with main rivers of South Africa. It was undertaken as part of the country's National Spatial Biodiversity Assessment (Driver et al., 2005) and is the first nation-wide assessment to apply similar approaches to concurrent assessments of terrestrial, marine and estuarine ecosystems, therefore facilitating comparisons across all four realms. There have been relatively few studies in South Africa dealing with systematic identification of rivers for conservation. Noble (1974) examined the representation of "aquatic biotopes" and habitats for threatened species in South Africa and on this basis derived an expert-based set of 23 aquatic sites for conservation. O'Keeffe et al. (1987) examined conservation status of selected rivers based on expert opinion of the relative importance of the river for conservation and the extent to which it had been disturbed from its natural state. These studies laid a good foundation for the criteria deemed important for conserving freshwater ecosystems. However, the study by Noble (1974) was not based on a systematic and spatially explicit classification of all freshwater ecosystems across the country; and the study by O'Keeffe et al. (1987) was a weighted scoring approach which ran the risk of undermining representation of ecosystems with a low conservation status, as is common for many scoring approaches (Pressey et al., 1994). It was only a decade later that the use of techniques based on principles of systematic conservation planning (Margules and Pressey, 2000) were applied in South Africa to identify landscape-level conservation priorities for rivers in the Cape Floristic

Region (Van Nieuwenhuizen and Day, 1999) and the Greater Addo Elephant National Park (Roux et al., 2002) Although these two studies were both systematic, focussing on achieving conservation targets for river biodiversity, as well as attending to important ecological and evolutionary processes which support and maintain this biodiversity in the long term, they were undertaken at a sub-national scale. There remained a need for a nation-wide systematic assessment of river conservation priorities, to provide context to water resource management and conservation activities in the country as a whole.

The results presented here serve as an initial step towards identifying systematic conservation priorities for rivers at a nation-wide scale. The short time-frame within which this assessment had to be completed (less than eight months) necessitated the development of a relatively rapid, pragmatic and inexpensive framework within which main river ecosystems were assessed. Both ecosystem status and protection levels of main river ecosystems were assessed. Ecosystem status is defined as a measure of the proportion of the river ecosystem still in its natural, intact state. Protection level of each river ecosystem is defined as the proportion of its minimum conservation target achieved in protected areas, where the minimum conservation target of each river ecosystem was set quantitatively as 20 % of its total length, and only intact river lengths contributed towards the target. This approach offers a new and relatively rapid framework for assessing river ecosystems. Species data, frequently a limiting factor in conservation assessments of river ecosystems, are not required. River integrity data are required; however, surrogates of river integrity can be applied where these data are limited (e.g. Stein et al., 2002). Endangerment categories thus generated allowed comparisons between terrestrial, river and marine ecosystems, whilst assessing protection levels in conjunction with river integrity offered a more meaningful method of measuring protected area gaps for river ecosystems.

METHODS

Defining main rivers

The 1:500 000 rivers data layer (DWAF, 2004a) was used in these analyses. Although this is based on 1:500 000 topographical maps, it has been refined to include alignment of the rivers to within 50 m of 1:50 000 topographical maps. Main rivers were defined using the South African Department of Water Affairs and Forestry quaternary catchments (Midgley et al., 1994). These catchments are part of a national hierarchical drainage sub-division system, which divides drainage regions into successively smaller hydrologic units: from primary catchments, through to secondary and tertiary catchments, and finally to quaternary catchments. This system is similar to the system used to delineate the U.S. Geological Survey (USGS) hydrologic units (Seaber et al., 1987), where quaternary catchments are comparable to the USGS cataloguing units. Main rivers were defined as the rivers which pass through a quaternary catchment into a neighbouring quaternary catchment (Figure 3.1). In instances where no river passed through the quaternary catchment (e.g. in coastal quaternary catchments which often encompass relatively short, whole river systems, or in quaternary catchments containing only endorheic rivers), the longest river system was chosen as the main river.

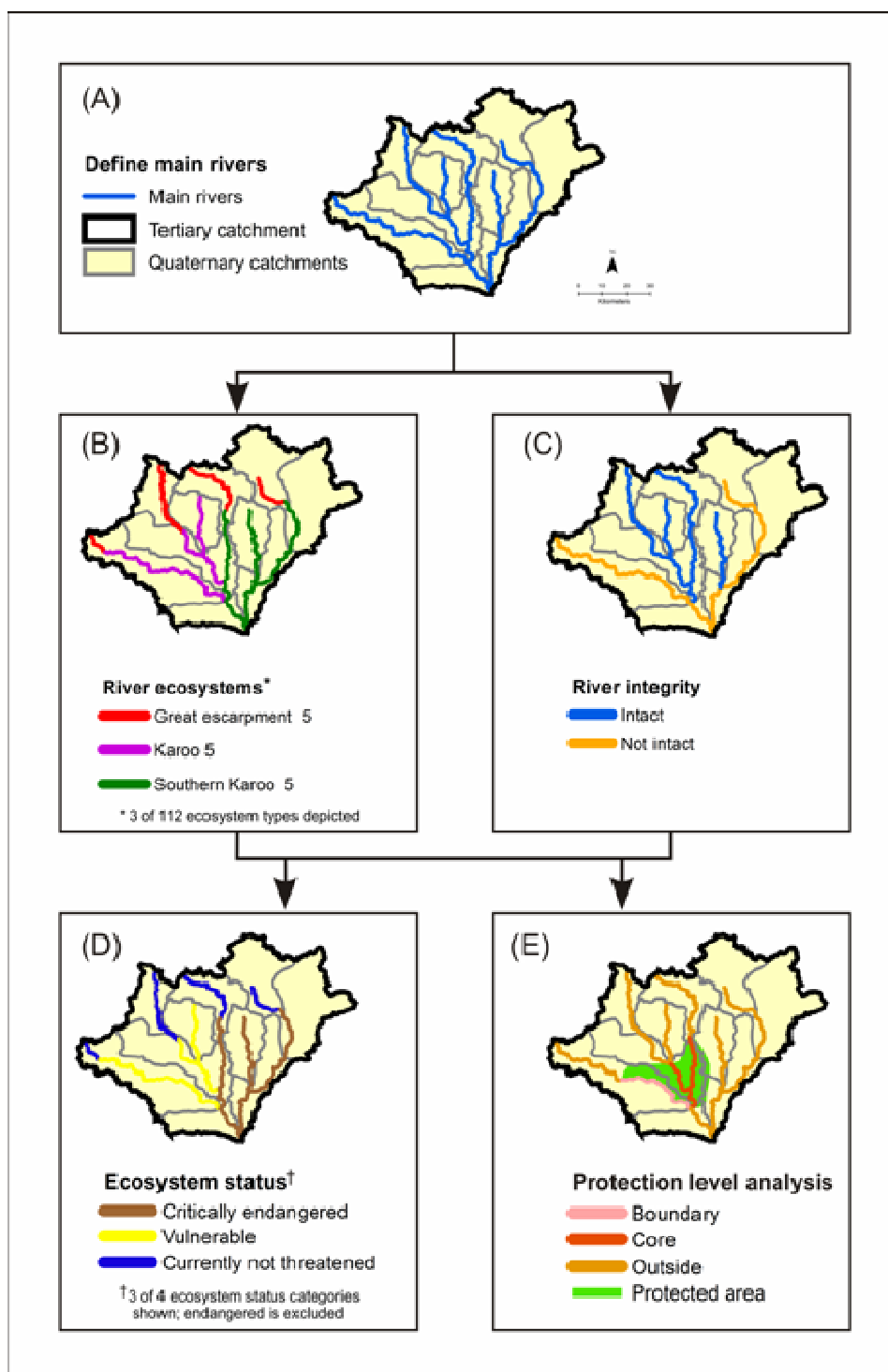


Figure 3.1 A schematic example of the steps used to derive ecosystem status and protection levels. Main rivers were defined using quaternary catchments (A). These main rivers were coded according to their ecosystem type (B) and river integrity (C). For each ecosystem type, the extent still intact (i.e. considered suitable for contributing towards quantitative conservation targets) was calculated, and ecosystem status was assigned using thresholds (D). Rivers were coded according to whether they fell outside protected areas (outside), formed the boundary of a protected area (boundary) or fell within a protected area (core). Intact core river lengths within statutory Type 1 protected areas were calculated for each ecosystem type (E), which was then assigned to an appropriate protection level category.

Mapping river ecosystems

River ecosystems were defined based on a hierarchical classification framework by Dollar et al. (2007). The framework characterises rivers according to geomorphological and hydrological descriptors, to derive components of rivers which, under natural conditions, are likely to share similar biological response potential, and can therefore be used as coarse-filter surrogates of river biodiversity (*sensu* Higgins et al., 2005). These components, hereafter “river ecosystems”, were derived for this national scale assessment by combining two spatial layers: geomorphic provinces (Partridge et al., 2006) as a freshwater-specific refinement of the provinces developed by King (1951), and hydrological index (Hannart and Hughes, 2003).

The 1:500 000 rivers layer was spatially overlaid with the layer for geomorphic provinces, to classify rivers according to the nature of the landscape through which it flows. Next, rivers were assigned a hydrological index class, broadly describing the amount and variability of water flow in a river. The hydrological index class for each river was derived by grouping hydrological indices at the quaternary catchment scale (Hannart and Hughes, 2003) into eight statistically derived classes (Dollar et al., 2006), where regions of low flow variability (commonly containing the perennial-type rivers) have a hydrological index class close to 1, and the semi-arid regions of high flow variability (commonly containing periodic- or ephemeral-type rivers) would be assigned to classes 6-8 (Table 3.1). Distinct combinations of geomorphic provinces and hydrological index classes assigned to rivers were used to depict river ecosystems at a national scale (Figure 3.1).

Table 3.1 Eight statistical classes of hydrological index derived using the hydrological indices of Hannart and Hughes (2003) for all 1986 quaternary catchments in South Africa, Lesotho and Swaziland. For South African rivers, regions of low variability (commonly containing the perennial-type rivers) have a hydrological index class close to 1, whilst semi-arid regions of high variability (commonly containing periodic- or ephemeral-type rivers) would be assigned to classes 6-8.

Hydrological index	Class
0 to 5	1
5.1 to 8	2
8.1 to 17	3
17.1 to 37	4
37.1 to 53	5
53.1 to 65	6
65.1 to 95	7
95.1 to 110	8

Mapping river integrity

Desktop estimates of present ecological status were used to depict river integrity in South Africa. These were developed for a national Water Situation Assessment Model (Kleynhans, 2000), where river integrity describes the extent to which the river has been modified by human activity (Kleynhans, 1996; Kleynhans, 1999). Estimates of river integrity were collected for the main rivers of all quaternary catchments through a series of local expert workshops throughout the country between 1998 and 1999. Six attributes (flow, inundation, water quality, stream bed condition, introduced instream biota, riparian or stream bank condition) were evaluated according to present ecological status categories ranging from A (natural) to F (critically modified). The six attributes were amalgamated into an overall estimate of instream and riparian habitat integrity, by calculating the median present ecological status category. For the purposes of this assessment, rivers with an overall present ecological status category of natural or largely natural (Class A or B respectively; see Table 3.2) were considered “intact” and suitable for contributing towards achievement of quantitative conservation targets. Targeting intact rivers for conservation maximizes the benefits already in place within these naturally functioning ecosystems. The median present ecological status category for each quaternary catchment main river was joined to the 1:500 000 main rivers GIS

layer, to provide a measure of integrity for each main river (Figure 3.1). An overview of the state of main river integrity in the country was calculated by summing the length of river reaches in each present ecological status category and expressing this as a percentage of the total length of main rivers in South Africa.

Table 3.2 State of main river integrity within South Africa, according to the desktop estimates of present ecological status categories (Kleynhans, 2000). Percentage of main river length was calculated by summing the length of river reaches in each present ecological status category and expressing this as a percentage of the total length of main rivers in South Africa. For the purposes of this study, rivers with a present ecological status of natural or largely natural (categories A or B respectively) were considered “intact”, and suitable for contributing towards quantitative conservation targets; categories C-F were considered unsuitable for contributing towards quantitative conservation targets.

Present ecological status category	Description as per Kleynhans (2000)	% Main river length
A	Natural, unmodified	4
B	Largely natural	25
C	Moderately modified	47
D	Largely modified	21
E to F	Seriously to critically modified	2

Ecosystem status

Main river ecosystems were combined spatially with the layer of river integrity to calculate the total intact length of each of the ecosystems associated with main rivers. The proportion of intact length to total length of each river ecosystem was measured to derive its ecosystem status (Figure 3.1). Ecosystem status was assessed based on thresholds which recognise minimum quantitative conservation targets for biodiversity pattern (below which an ecosystem becomes *critically endangered*); and thresholds which recognise conservation targets for maintaining ecological and evolutionary processes that sustain biodiversity pattern and allow it to evolve naturally over time (which in turn determine whether an ecosystem is *endangered*, *vulnerable* or *currently not threatened*). Setting thresholds is a potentially valuable concept to use as a basis for developing tools to conserve and manage biodiversity. However there are a range of

uncertainties in the application of thresholds (Huggett, 2005), and future empirical studies are required to support the thresholds used in this study.

The minimum conservation target, as described in Margules and Pressey (2000), was set for each river ecosystem as 20 % of its total river length. This 20 % target is a value endorsed by key government departments responsible for conserving freshwater ecosystems in South Africa (Roux et al., 2006). *Critically endangered* river ecosystems have an intact length < 20 % of their total original extent (i.e. their minimum conservation target). Dropping below this threshold implies that the ecosystem is inadequately represented in the country, and has become critically endangered. *Endangered* river ecosystems have an intact length < 40 % and ≥ 20 % of their total length; these ecosystems have lost significant amounts of their natural habitat, and their ability to support ecological and evolutionary processes is likely to be compromised. *Vulnerable* river ecosystems have an intact length < 60 % and ≥ 40 % of their total length; these ecosystems have lost some of their original natural habitat, and their ability to support ecological and evolutionary processes is likely to be compromised if they continue to lose natural habitat. River ecosystems classified as *currently not threatened* have an intact length ≥ 60 % of their total length; these systems have lost a smaller proportion of original habitat. Ecosystems with a status of critically endangered, endangered and vulnerable were considered threatened ecosystems in this assessment. It is acknowledged that thresholds used to identify threatened ecosystems are oversimplified and should be refined as better empirical data and methods for target setting become available.

The spatial distribution of ecosystem status was examined within the context of flow predictability by comparing the ecosystem status in more permanently flowing main rivers (defined as those with a hydrological index class of 1-5, Table 3.1) with those main rivers that have a more variable flow (defined as those with a hydrological index class of 6-8, Table 3.1).

Protection levels

River ecosystems were spatially combined with a layer of protected areas compiled for the terrestrial national spatial biodiversity assessment (Reyers et al., 2007), to calculate the proportion of each river ecosystem currently under formal protection. Only statutory Type 1 protected areas (77 % of the mapped protected areas) were used in these analyses, which include National Parks, Provincial Nature Reserves, Local Authority Nature Reserves and Department of Water Affairs and Forestry Nature Reserves. The remaining protected areas (Types 2 and 3) have not been comprehensively mapped and legislation governing these protected areas is less certain (Driver et al., 2005). Since many rivers form boundaries of protected areas, a distinction was made between boundary rivers that are protected on one side only, and core rivers that are protected on both sides of their river bank. Boundary rivers, defined as those rivers that fell within a buffer of 500 m either side of the protected area boundary, were excluded from these analyses. Any core rivers that were not intact were also excluded, i.e. only intact core river lengths within statutory Type 1 protected areas were used in these analyses (Figure 3.1).

River ecosystems were assigned a protection level based on the percentage of their minimum conservation target (20 % of their total length) achieved by intact core river lengths within statutory Type 1 protected areas, as follows: not protected (0 %), hardly protected (<5 %), poorly protected (5-50 %), moderately protected (50-99.9 %) and well protected (\geq 100 %).

RESULTS

Main river ecosystems and their integrity

Main rivers, as defined in this assessment, constitute less than 45% of the rivers analysed at the 1:500 000 scale; the remainder are considered tributaries. There are 112 river ecosystems associated with main rivers, defined using distinct combinations of geomorphic provinces and hydrological index classes (Figure 3.2, Appendix 1). Four of

these river ecosystems occur only in main rivers, i.e. there are no examples of these river ecosystems contained in tributaries.

According to the estimates of present ecological status (Kleynhans, 2000), less than a third of main rivers in South Africa are still intact and suitable for contributing towards minimum conservation targets (Table 3.2). The majority of main rivers (47 %) are moderately modified, whilst 23 % of them can be considered irreversibly transformed in terms of their ability to support biodiversity, and are deemed unsuitable for conservation (those rivers that fall into the D, E or F present ecological status categories, Table 3.2).

Ecosystem status

An alarming 84 % of South Africa's 112 main river ecosystems are threatened (Figure 3.3a, Appendix 1), with 54 % critically endangered, 18 % endangered, and 12 % vulnerable. Only 16 % of main river ecosystems are currently not threatened. The more permanently flowing main rivers have a higher proportion of threatened ecosystems than those main rivers with variable flow (Figure 3.3b). The semi-arid interior of the country, characterised by rivers with variable flow, is therefore the only area in South Africa that still contains a large proportion of main river ecosystems that are currently not threatened (Figure 3.4). Main rivers in the rest of the country contain mostly threatened ecosystems, except in the vicinity of the larger protected areas (Figure 3.4).

Two of the four river ecosystems that are unique to main rivers are critically endangered (Lower Vaal and Orange valleys-5 and Swartland-5; Appendix 1). For these ecosystems, there are no tributaries that could contribute towards their conservation. However, for the rest of the critically endangered main river ecosystems, options may exist for their conservation in intact tributaries, which, in general are less impacted than main rivers.

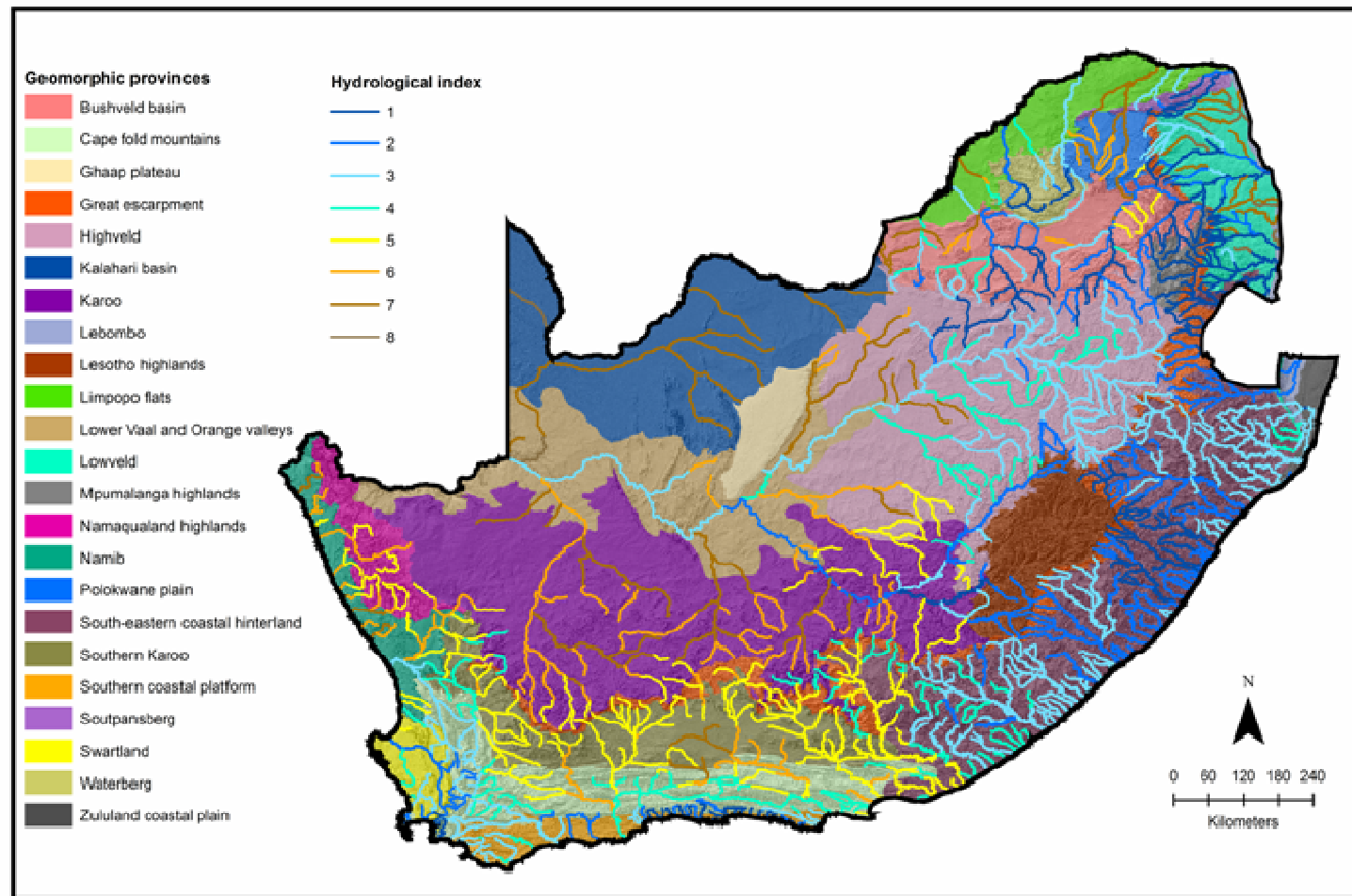


Figure 3.2 Main river ecosystems in South Africa (n = 112). River ecosystems were defined using unique combinations of geomorphic province (shaded areas) and hydrological index class (coloured lines).

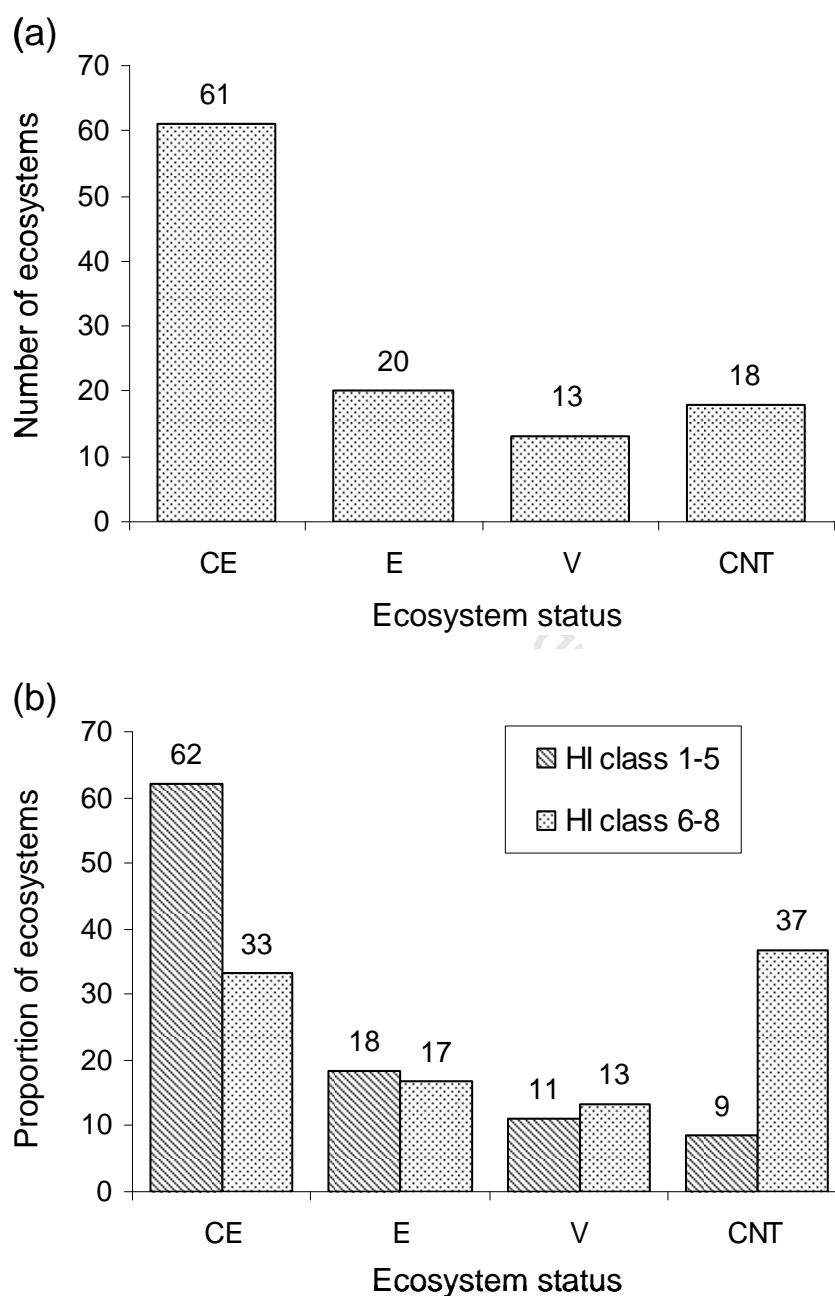


Figure 3.3 (a) The number of main river ecosystems ($n = 112$) that are critically endangered (CE; $< 20\%$ intact), endangered (E; $20-40\%$ intact), vulnerable (V; $40-60\%$ intact) and currently not threatened (CNT; $> 60\%$ intact); and (b) ecosystem status of the more permanently flowing main rivers compared to that of rivers whose flows are more variable, where rivers with a hydrological index (HI) class of 1-5 are considered more permanent and those with a HI class of 6-8 more variable. Proportion of ecosystems is calculated as the number of ecosystems expressed as a percentage of the total number of ecosystems in each group.

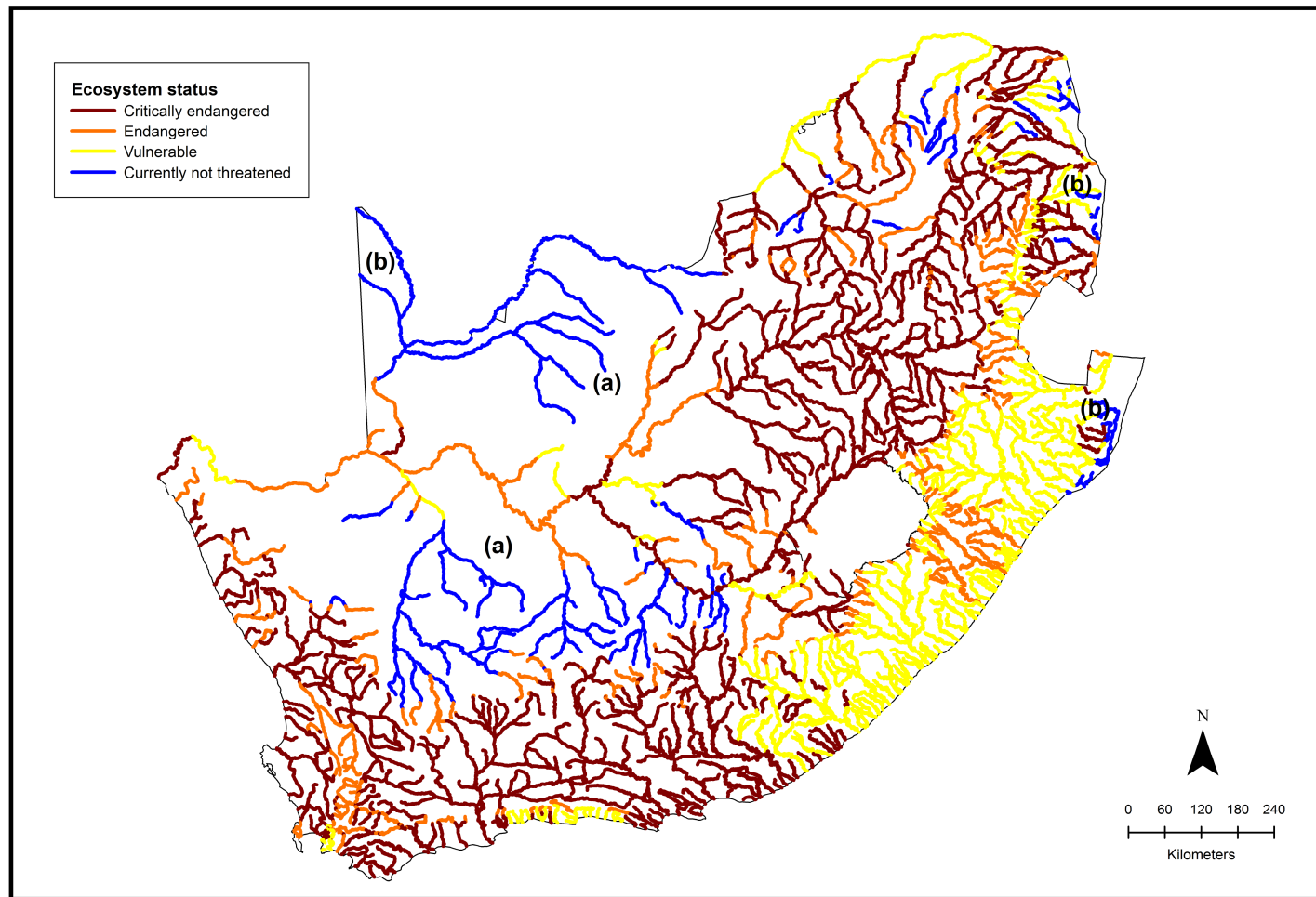


Figure 3.4 Ecosystem status of main rivers in South Africa, based on the extent of ecosystem still intact. All main rivers are depicted according to their ecosystem status at a national scale, i.e. if a river contains a critically endangered ecosystem, that portion of the river is depicted as critically endangered, regardless of its ecological integrity. The approximate vicinities of the arid interior and larger protected areas, referred to in the text, are denoted by (a) and (b) respectively.

Protection levels

Over 90 % of all main rivers in South Africa fall completely outside statutory Type 1 protected areas (Table 3.3). Half of the remaining rivers form boundaries of protected areas; thus less than 5 % of main rivers in the country are core rivers within protected areas, receiving protection on both sides of their river bank. Just over 50 % of the core river length is still intact, showing an improvement in overall condition compared to rivers falling completely outside of protected areas, which have only 28 % of their river length still intact. As could be expected, rivers forming the boundaries of protected areas have an overall condition that is lower than core rivers, but better than rivers completely outside protected areas (Table 3.3).

Table 3.3 Proportion of main rivers in South Africa falling outside protected areas (Outside), on boundaries of protected areas (Boundary) or within protected areas (Core, i.e. > 500 m from boundary). Proportion of river length still intact is also given.

Location of river	% Total length in South Africa	% Length intact
Outside	92	28
Boundary	4	36
Core	4	51

Sixty-five of the 112 (58 %) main river ecosystems are either not protected, or have no remaining intact length (Figure 3.5). A further 31 main river ecosystems receive low levels of protection. Only 16 (14 %) main river ecosystems are moderately to well protected, having achieved over half of their minimum conservation target (i.e. > 10 % of their total length) in statutory Type 1 protected areas.

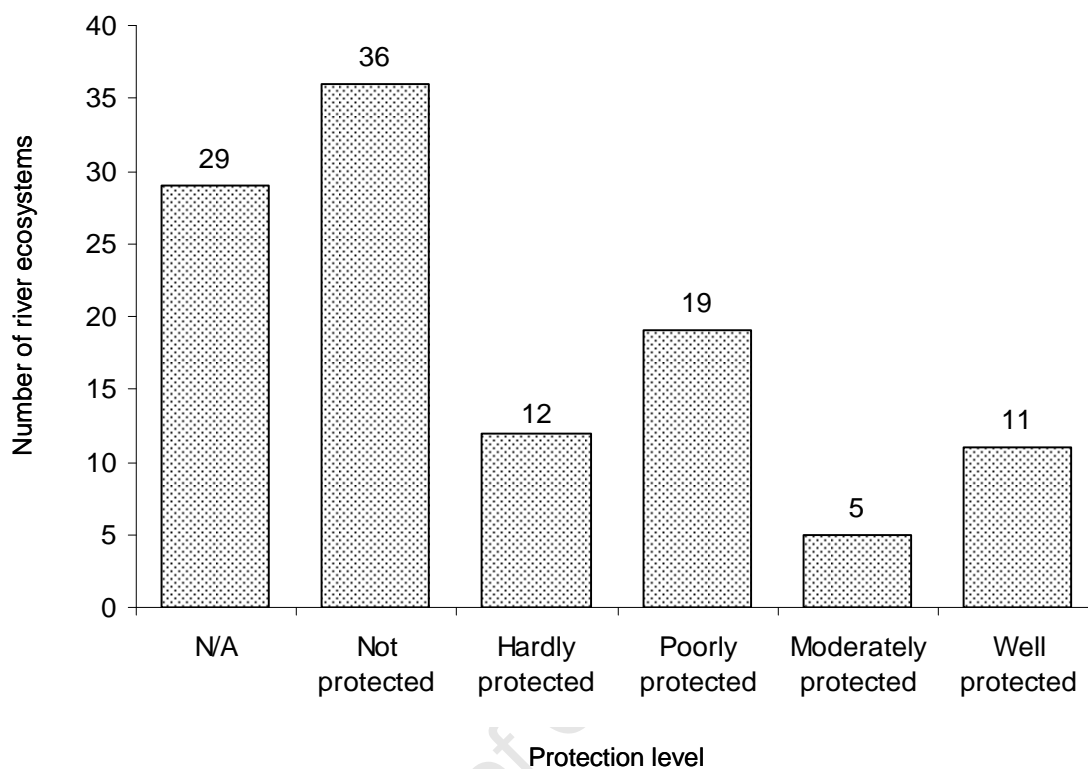


Figure 3.5 The number of river ecosystems ($n = 112$) that are Not protected (0 %), Hardly protected (<5 %), Poorly protected (5-50 %), Moderately protected (50-99.9%) and Well protected ($\geq 100\%$) within statutory Type 1 protected areas. Protection levels are based on the proportion of quantitative conservation target met within protected areas, where the conservation target was taken as 20 % of the total length of each main river ecosystem in South Africa. "N/A" represents those river ecosystems that were not applicable to this analysis because they had no intact main river remaining. Only intact rivers falling within protected areas and > 500 m from boundary, as opposed to forming the boundary, were considered as contributing towards this conservation target.

DISCUSSION

Ecosystem status

This assessment applied a similar approach to parallel assessments of terrestrial, marine and estuarine ecosystems, and it revealed that main river ecosystems are in a critical state, far worse than terrestrial ecosystems: 54 % of main river ecosystems are critically endangered, compared to the 5 % of critically endangered terrestrial ecosystems (Driver

et al., 2005; Reyers et al., 2007). These results mimic published literature on global trends of the state of freshwater and terrestrial biodiversity (McAllister et al., 1997; Ricciardi and Rasmussen, 1999; Abell, 2002; Higgins, 2003; Gleick, 2004; WWF, 2004). The alarming state of main river ecosystems has important implications in developing strategic government direction and policy concerning biodiversity conservation in the country. Freshwater needs to be placed at the forefront of biodiversity planning and implementation (e.g. in the National Biodiversity Strategy and Action Plan), to ensure conservation of freshwater ecosystems and the important services they provide.

Main rivers in South Africa are heavily utilized and regulated to improve water security for socio-economic use, and there are widespread water transfer schemes across the country to cater for areas where water requirements exceed the natural water availability (Braune, 1985; O’Keeffe, 1989; DWAF, 2004b). This places a great deal of stress on natural ecosystems, as demonstrated by the ecosystem status assessment which shows that 84 % of main rivers have become degraded to the point at which they are now threatened (Figure 3.3a). Furthermore, the more permanently flowing main rivers, which tend to lend themselves better to utilisation and regulation than those rivers with more variable flow, have a higher proportion of threatened ecosystems (Figure 3.3b). Modifications to perennial rivers are often associated with significant investments in infrastructure and development (e.g. construction of large dams and irrigation schemes), which makes remedial action difficult from political and socio-economic perspectives.

This assessment is based on main rivers only, and ignores the conservation potential of the numerous major tributaries feeding the main rivers, which are often representative of the same types of ecosystems and in better condition. Had tributaries been included in this assessment, some river ecosystems shared between main rivers and tributaries may well have been classified as less threatened. This highlights the importance of tributaries for conserving biodiversity, in which conserved tributaries could be viewed as refugia for river biodiversity, replenishing other parts of the river system from time to time. For this replenishment to occur, however, it is important that the longitudinal connectivity between the tributaries and its main river be maintained.

From a management perspective, a multiple-use landscape, which seeks to balance the needs of resource utilization and biodiversity conservation, is therefore proposed. In this management scenario, intact tributaries would play a crucial role in meeting conservation targets, and these would need to be maintained in a relatively natural state with no discharges or impoundments. Main rivers of tributaries selected for conservation could be moderately utilized but would need to be maintained in a healthy enough state to facilitate longitudinal connectivity; this requires understanding ecological needs and designing dam releases accordingly (Postel and Richter, 2003). This supports global findings that conserving biodiversity and meeting human needs does not have to be mutually exclusive (Gilman et al., 2004; Richter et al., 2003).

As an initial step towards prioritising conservation action, it is recommended that conservation attention be focussed on conserving intact tributaries containing critically endangered main river ecosystems, whilst maintaining main rivers in a state healthy enough to facilitate longitudinal connectivity between conserved tributaries. Conservation of the two critically endangered river ecosystems that are unique to main rivers (the Lower Vaal and Orange valleys-5 and the Swartland-5; Appendix 1) cannot be supported by tributaries. If minimum conservation targets are to be met for these ecosystems, portions of suitable main river will need to be rehabilitated. If this is not possible (e.g. owing to socio-economic constraints), this assessment at least makes explicit exactly which ecosystems we have lost or would lose, thus enabling an examination of the subsequent consequences.

Protection levels

Globally, as in South Africa, there has been very little emphasis on proclaiming protected areas for the primary purpose of conserving freshwater ecosystems (Saunders et al., 2002). It is therefore not surprising that most main river ecosystems are not represented in protected areas (Figure 3.5). Moreover, inclusion in protected areas does not guarantee conservation: only 50 % of the core rivers within protected areas are intact (Table 3.3). In extreme cases, rivers within protected areas are considerably degraded because they are designed around dams; in most cases, rivers are inadequately conserved because they are not fully contained within protected areas, and are

negatively impacted by activities outside the protected area, such as dam construction and agriculture. Despite these deficiencies, the higher proportion of intact rivers inside protected areas, compared to outside (Table 3.3) emphasizes the positive role protected areas can have, through appropriate land management strategies.

Saunders et al. (2002) provide a few examples where protected areas have been designed with the conservation of freshwater in mind. However, they recognise that whole catchment protection is often difficult to attain, and put forward alternative freshwater protected area design and management strategies, including application of multiple-use zones, use of vegetated buffer strips, attention to ecological flow requirements and eradication of exotic species. Whilst conserving whole river systems in protected areas is seldom a practical management option, changing the way in which future protected areas are designated or expanded could help improve the representation of freshwater ecosystems within protected area systems. These include (i) giving explicit consideration to representing freshwater ecosystems in protected areas (Chapter 4); (ii) understanding the relative contribution different land makes to freshwater conservation in consolidating land around existing protected areas (Chapter 4; Roux et al., 2002); (iii) avoiding the use of rivers to delineate boundaries of protected areas; and (iv) using alternative design and management strategies (Chapter 6; Saunders et al., 2002), in combination with existing protected areas, to protect rivers before they enter the protected area.

Although more attention needs to be given to conserving freshwater biodiversity in formal protected areas, this management option alone is not feasible for meeting conservation targets of all ecosystems (currently only 14 % of main river ecosystems are moderately to well protected). The most feasible management solution is one of integrated river basin management (IRBM) within catchments, which takes into account the interrelationships between water, the biophysical environment, and socio-economic and political factors. However, Gilman et al. (2004) have found that systematic conservation planning for freshwater biodiversity is underrepresented in most IRBM plans, particularly in developing countries. There is thus an urgent need for promoting the systematic and purposeful conservation of freshwater biodiversity within the context of most IRBM programmes. IRBM plans need to develop clear and explicit

conservation visions, targets and guidelines to ensure the sustainability of freshwater ecosystems and their services, even as stakeholder interests in the area develop. In South Africa, the national Department of Water Affairs and Forestry, custodians of the country's water resources, have acknowledged this need through a project aimed to develop cross-sectoral policy objectives for inclusion of systematic conservation of freshwater ecosystems in their strategic planning processes (Roux et al., 2006).

Limitations and future improvements

Distinguishing between main rivers and tributaries was useful in highlighting the dire state of main rivers and their ecosystems in the country. However, an assessment of both main rivers and tributaries will give a more complete picture of overall ecosystem status of rivers in the country, and the ability to achieve longitudinal connectivity across the landscape. This is currently not possible owing to the lack of data on ecological integrity of tributaries at a national scale. The main river integrity data are also outdated, with transformation having proceeded at alarming rates since the derivation of these data. There is thus a need to update the national scale river integrity data to include both main rivers and major tributaries (Chapter 4). This updating should take cognisance of the numerous sub-national river health surveys (e.g. RHP, 2001a; RHP, 2001b; Chapter 6).

Lack of available data on river ecological integrity in Lesotho, Swaziland and Mozambique also prevented an assessment of ecosystems associated with rivers shared by neighbouring countries. Assessing river basins that are not split by political boundaries would provide a more complete, regional assessment of ecosystem status, highlighting ecosystems whose conservation requires the cooperation of more than one country. Nevertheless, this assessment was useful for informing national policy-makers of the status of freshwater ecosystems at a country-wide scale.

River ecosystems used in these analyses are in the process of refinement, and should therefore be viewed as preliminary. Once the ecosystems have been refined, they need to be reviewed by experts to assess whether they provide a true reflection of river ecosystem types at a national scale. The adequacy of these river ecosystems as biodiversity surrogates in conservation planning should also be tested.

There is a range of uncertainty in the setting of thresholds used for devising the different ecosystem status categories. These include issues such as ability to identify ecological thresholds, the variation in the response of different species or ecosystems to the same disturbances and the variation in response to thresholds at different scales (Huggett, 2005). There is a strong need for empirical data to support the thresholds between the ecosystem status categories (20 %, 40 % and 60 %); these studies would improve the scientific understanding of river ecology, ecosystem functioning and the response of ecological variables to disturbances. In addition, uniform thresholds undermine the relative responses of different ecosystems to the same disturbances. Thresholds used in this assessment should therefore be refined as new research becomes available.

This assessment was considerably limited in drawing conclusions about the prioritisation of rivers for conservation action because it did not include an assessment of tributaries, and was unable to examine the vulnerability of different rivers to future threats. As a first step in prioritisation, conservation action could focus on healthy tributaries containing critically endangered main river ecosystems. Future refinements of this study should focus on developing a more robust priority layer that includes both an analysis of the contribution tributaries make to conservation targets (Chapter 4), as well as an analysis of vulnerability to future threats. Apart from extending this assessment to include tributaries, there is an additional need to consider wetlands and ground water, as well as to include an assessment of key species or species groups (Chapters 4 and 6).

CONCLUSIONS

This study provides an assessment that examines endangerment and protection levels of rivers within large catchments, at a scale appropriate for informing conservation action at a national level (quaternary catchments). The results produced were systematic, defensible and alarming, confirming general suspicions of the state of main river ecosystems. One of the main advantages of this assessment is that the results were used to guide the National Biodiversity Strategy and Action Plan, which has a strong focus on implementation (Driver et al., 2005). Figures were therefore designed to create a

visual impact for decision makers, and undertaking this assessment with concurrent assessments of terrestrial, marine and estuarine ecosystems also drew attention to the strategic national need to pay more attention to the state of freshwater biodiversity (Driver et al., 2005).

As demands on water increase, the impounding of main river flows to provide water security is likely to increase. This study highlights the importance of intact tributaries for achieving river conservation targets, since tributaries are generally less regulated than main rivers. However, this does not preclude the need for managing main rivers as conduits across the landscape to support ecological processes that depend on connectivity. In management terms, a moderately used main river connecting intact tributaries may be the best means of achieving a balance between resource utilization and resource protection, particularly in water-limited countries.

Whilst protected areas do not adequately protect river ecosystems assessed in this study, there is a marked improvement in overall river integrity inside protected areas compared to outside. This provides a strong, quantitative argument for establishing protected areas that target freshwater ecosystems, species, and the functional processes that support these (Chapter 4). This can be initiated by expanding existing protected areas where possible to include whole river systems, avoiding the use of rivers to delineate boundaries of protected areas and attempting to conserve entire catchments. Where inclusion of entire catchments is not feasible, an attempt should be made to protect rivers before they enter protected areas through the application of management strategies such as delineation of multiple-use zones, riparian zones, and partial water discharges in line with ecological flow requirements.

River conservation is entirely dependent on sound management of the entire catchment they drain. They therefore rely on effective IRBM and there is an urgent need for IRBM plans to include explicit conservation visions, conservation targets and guidelines to ensure that the needs of freshwater biodiversity are met, even as stakeholder needs grow. This will also ensure the sustainable provisioning of ecosystem services derived from freshwater ecosystems.

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CHAPTER 4.
EXPANDING PROTECTED AREAS BEYOND
THEIR TERRESTRIAL COMFORT ZONE:
IDENTIFYING SPATIAL OPTIONS FOR
RIVER CONSERVATION

University of Cape Town

EXPANDING PROTECTED AREAS BEYOND THEIR TERRESTRIAL COMFORT ZONE: IDENTIFYING SPATIAL OPTIONS FOR RIVER CONSERVATION

ABSTRACT

There has been very little consideration of freshwater ecosystems in identifying and designing protected areas. It is therefore not surprising that protected area systems worldwide show significant gaps in their conservation of freshwater biodiversity. Recent studies suggest that protected areas hold enormous potential to conserve freshwater biodiversity if augmented with appropriate planning and management strategies. Recognizing this need, South Africa's relevant government authority commissioned a spatial assessment to inform their national protected area expansion strategy. This study presents the freshwater component of the spatial assessment, aimed at identifying focus areas for expanding the national protected area system for the benefit of river biodiversity. It begins by establishing a set of conservation objectives to guide the assessment. These objectives seek to improve representation of river biodiversity pattern and processes in both new and existing protected areas. Data to address these objectives were collated in a Geographic Information System (GIS) and a conservation planning algorithm was used as a means of integrating the multiple objectives in a spatially efficient manner. Representation of biodiversity pattern was based on achieving conservation targets for 222 river types and 47 freshwater fish endemic to South Africa. Options were also identified for representing large-scale biodiversity processes associated with free-flowing rivers and catchment-estuarine linkages. River reaches that with only minor expansion of existing protected area boundaries could be fully incorporated into the national protected area system were also identified. This study concludes with recommendations for designing protected area systems, discussed in the context of planning for river biodiversity.

KEY WORDS: Systematic conservation planning, gap analysis; conservation assessment; free-flowing rivers; estuary; catchment to coast

INTRODUCTION

Around the world, governments have made commitments to establish protected area systems that contain viable representations of every terrestrial, freshwater and marine ecosystem (IUCN, 2003). However, several recent studies worldwide have highlighted significant gaps in protected area systems for freshwater ecosystems, both in terms of their representation and their ecological viability and integrity (Chapter 3; Keith, 2000; Yip et al., 2004; Abellán et al., 2007).

There are at least three reasons for this. First, there has been very little emphasis on freshwater ecosystems in identifying and designing protected areas – they are generally only protected incidentally through their incorporation into terrestrial protected areas (Saunders et al., 2002). Second, protected area management has focussed largely on managing terrestrial biodiversity – in many instances freshwater ecosystems within protected areas have even been deliberately altered by the construction of dams, roads, bridges and tourist lodges (Gaylard et al., 2003). Third, partial inclusion of rivers in protected areas is no guarantee for their protection since impacts outside protected area boundaries can still have negative consequences for freshwater biodiversity within them (Mancini et al., 2005). This means that protected area management plans need to acknowledge processes and threats external to their boundaries.

Consistent with the international trend, South Africa's system of protected areas shows significant gaps in conserving freshwater ecosystems. A recent conservation assessment of large river systems in South Africa (Chapter 3) found that: (1) representation of river ecosystems in protected areas is alarmingly inadequate; (2) almost half of the large river systems that are incorporated into protected areas are not intact, having been degraded by upstream human activities before entering the protected area; and (3) half of the river systems associated with protected areas are used to delineate boundaries and therefore only enjoy the benefit of protected area management on one side of their banks, if at all.

An important and more optimistic finding stemming from this study was that at a national scale, river systems in protected areas appear to be in a better overall

condition than those outside of protected areas. This emphasizes the positive role protected areas can play in conserving freshwater ecosystems and associated biota. However, realizing the full potential of protected areas in conserving freshwater ecosystems will require explicit incorporation of freshwater biodiversity into both protected area planning and management (Roux et al., 2008).

In South Africa, the opportunity arose to incorporate freshwater biodiversity into spatial planning for protected areas when the national Department of Environmental Affairs and Tourism (DEAT) commissioned the development of a strategy to guide the expansion of the country's land-based protected area system – including both the establishment of new protected areas and expansion of existing ones. As input into the strategy, a spatial assessment of both terrestrial and freshwater biodiversity was undertaken to identify focus areas for expanding protected area systems for the benefit of both realms. This study presents the freshwater component, focussing on rivers as an initial step, with a view to expanding to a broader suite of freshwater ecosystems over time. The study begins by outlining multiple conservation objectives to guide such analyses, and then demonstrates how these objectives can be tackled and integrated using a systematic conservation planning algorithm. Finally, generic recommendations are made regarding how to locate, design and manage land-based protected areas so as to improve the potential of protected area systems for river biodiversity. These recommendations also discuss the potential options for integrating this assessment with the concurrent spatial assessment of terrestrial biodiversity.

METHODS

Objectives to guide analyses

Conservation objectives guiding this assessment included the representation of river biodiversity pattern (e.g. fish species and river types) and processes (e.g. free-flowing rivers) in both new and existing protected areas. In addition, strategic opportunities were identified for improving the persistence of river biodiversity through minor expansion of existing protected areas (Table 4.1).

Table 4.1 Conservation objectives used to guide identification of freshwater focus areas for expanding protected area systems.

Objective	Rationale
1. Improve overall representation of river types and freshwater fish species endemic to South Africa, particularly threatened river types	River types and freshwater fish species serve as a coarse-fine surrogate approach to conserving representative examples of river biodiversity in South Africa. Threatened river types are particularly targeted since limited options remain for their conservation. Here, threatened river types are defined using the endangerment categories of Nel et al. (2007), which are based on the proportion of total length of that river type still intact.
2. Select intact river systems	These systems are the ones that are most likely to support ecologically viable biodiversity components in the long term.
3. Promote new protected areas for conserving the remaining free-flowing rivers	Conserves representative large-scale processes such as natural flow regimes, erosion and sediment transport. There are very few free-flowing rivers left in South Africa.
4. Represent intact rivers connected to priority estuaries	Conserves representative examples of catchment-scale processes that link land, water and sea.
5. Identify ecologically functional river reaches that could be fully incorporated into a protected area with only minor expansion	Highlights potential opportunities for strengthening the persistence of rivers in existing protected areas. These opportunities should be investigated further in terms of practical and ecological feasibility.

River types were used as a coarse-filter surrogate for representation of biodiversity pattern. Coarse-filter surrogates focus representation on higher levels of the biodiversity hierarchy, such as ecosystems, assuming that this will also conserve many associated species, communities and ecological processes (Hunter, 1991). River types were supplemented with distributional data on freshwater fish endemic to South Africa, which served as a fine-filter surrogate. Freshwater fish endemics were chosen because these are often the species that fall through the coarse-filter net (Lombard et al., 2003), and loss of these species would be globally significant.

Two issues were considered in terms of representing biodiversity processes (Table 4.1): representing the last remaining free-flowing rivers, and representing linkages between intact river systems and priority estuaries. A third related issue, that of improving the persistence of river biodiversity in existing protected areas, focussed on identifying opportunities where minimal expansion of existing protected area boundaries would enable the full incorporation of river reaches that are currently only partially protected.

River network and sub-catchments

This study was based on the 1:500 000 rivers Geographic Information System (GIS) layer within the boundaries of South Africa, Lesotho and Swaziland (DWAF, 2006). This GIS layer is at a fine enough scale to draw pragmatic decisions about national water resource management, and has been refined to include alignment of the rivers to within 50 m of 1:50 000 topographical maps (Department of Land Affairs, 2005). Because available river integrity data exist for main rivers only, a distinction was made between 1:500 000 main rivers and tributaries. Main rivers were defined according to Chapter 3 using the South African quaternary catchments, which are nested hydrologic units within primary, secondary and tertiary catchments (Midgley et al., 1994). Main rivers span more than one quaternary catchment, while tributaries are completely contained within single quaternary catchments.

The 1:500 000 rivers GIS layer also includes a river network typology, where river segments between confluences are assigned a unique identifier that allows rivers upstream and downstream to be identified and grouped. This typology was used for GIS

analyses of representation and persistence. In considering representation of biodiversity pattern, the assessment was conducted at the scale of a river segment, defined as the portion of river between confluences of the 1:500 000 rivers GIS layer (Figure 4.1). For considerations of connectivity and persistence, assessments were undertaken at the scale of a 1:500 000 river reach, defined as a whole river sub-system from its headwaters to either the estuary or confluence with a major river (Figure 4.1). A river reach can be made up of several river segments, and may be relatively short or as long as, e.g., the Gariep River (almost 2 000 km).

Sub-catchments were used as planning units in deriving spatial focus areas for expanding protected area systems. These were modelled for the whole country around each river segment in GIS (Arc Hydro, Version 1.1, ESRI, Redlands, CA) using 90 m resolution digital elevation data (US Shuttle Radar Topography Mission data; <http://srtm.usgs.gov/>).

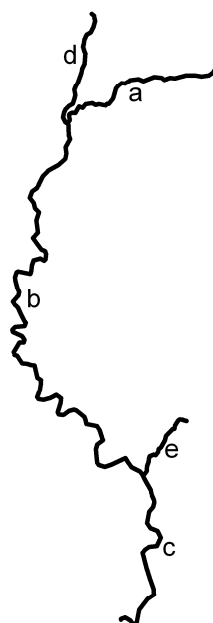


Figure 4.1 Difference between river segments and river reaches. Five river segments are shown between river confluences, labelled a to e. These make up three river reaches – one comprised of multiple river segments a to c; and the remaining represented by d and e. Sub-catchments were delineated around each river segment.

River integrity

Ecological integrity of all 1:500 000 rivers was mapped using existing data for main rivers in combination with a modelling approach for the tributaries. The ecological integrity of main rivers was described using the categories from Kleynhans (2000), where rivers with an overall category A or B were considered “intact” (Chapter 3). Owing to a lack of comprehensive data, ecological integrity for tributaries needed to be modelled using the percentage of natural land cover as a surrogate. Remotely-sensed land cover data are the most common surrogate measures used to infer information about the impact that human activities have on freshwater systems (Stein et al., 2002; Linke et al., 2007; Norris et al., 2007; Thieme et al., 2007). Only two categories of integrity were assigned to tributaries: “intact” (equated to the A or B ecological integrity categories of main rivers), or “not intact” (assigned to a category of Z). Natural and transformed land cover classes were defined from the 30 m resolution South African National Land Cover 2000 GIS layer (Fairbanks et al., 2000). Transformed land classes included cultivated, urban, degraded and eroded land, as well as plantations, mines and quarries. Farm dams at a 1:50 000 scale (Department of Land Affairs: Chief Directorate of Surveys and Mapping, 2005) were also used to distinguish man-made and natural waterbodies. The remaining land cover classes were considered natural.

Four steps were used to calculate ecological integrity categories for each river segment (Figure 4.1). First, the percentage of natural land cover was calculated for each river segment within its sub-catchment, and within a 500 m and 100 m GIS buffer of the river segment. Second, the minimum of these three percentages was assigned to each river segment. Third, any river segment with a minimum natural land cover of $\geq 75\%$ was assumed to be “intact” (i.e. ecological integrity category A or B); any river segment below this threshold was taken as “not intact” (assigned an ecological integrity category of Z). This threshold was guided by expert knowledge and comparisons of some of the site-assessment data with modelled outputs. Finally, any intact river segment with $\geq 5\%$ erosion within a 500 m GIS buffer of the river reach was downgraded to “not intact”. This was done to account for the inaccuracy of the land cover data in detecting land degradation, which is a problem in drier areas where subsistence grazing often causes

disproportionate degradation to rivers, altering the riparian vegetation and causing bank erosion (Thompson et al., 2008).

The level of confidence in the river integrity data is higher for main rivers than for tributaries, because main river data has been through a process of expert review. This was an important distinction in identifying qualifying river systems for addressing objectives of Table 4.1. Intactness of both main rivers and tributaries was used as a qualifying criterion for representing examples of biodiversity pattern and processes. For consideration of connectivity, where less stringent rules of intactness were applied, integrity of tributaries was disregarded because of the lower confidence in these data.

Biodiversity pattern

Three GIS layers were combined to derive river types for the 1:500 000 river network. First, each river segment was classified according to the majority Level 1 ecoregion (Kleynhans et al., 2005) through which it flowed. These river ecoregions are based on the hierarchical ecoregional typing approach of Omernik (1987) and characterise regions within which there is relative similarity in the mosaic of ecosystems and ecosystem components (biotic and abiotic, aquatic and terrestrial). Next, river flow variability was described as either “permanent” or “not permanent” using 1:50 000 topographical maps (Department of Land Affairs, 2005), where “permanent” groups perennial and seasonal rivers, and “not permanent” refers to ephemeral rivers. The third GIS layer consisted of geomorphological zones derived for all 1:500 000 river channels (Moolman et al., 2006) using descriptions and slope categories proposed by Rowntree and Wadson (1999). The seven geomorphological zones thus identified were grouped into four ecological classes: mountain streams, upper foothills, lower foothills and lowland rivers.

A list of the endemic freshwater fish species was provided by the South African Institute of Aquatic Biodiversity (SAIAB). This list includes 47 endemic freshwater fish in South Africa, comprised of both currently described species and evolutionary significant units, which are distinct populations within species, described on the basis of ecological and genetic data (Moritz, 1994). Distribution records for these endemic freshwater fish were extracted from the SAIAB and Albany Museum fish databases.

Biodiversity processes

River reaches satisfying all of the following requirements were selected as free-flowing rivers: (1) permanent or seasonally flowing; (2) intact; (3) no instream dam throughout its length; and (4) length ≥ 50 km for inland rivers, with no size threshold for coastal rivers. River type surrogates and river integrity were used to identify reaches qualifying under (1) and (2) respectively. The 1:50 000 farm dams (Department of Land Affairs, 2005) were used to identify instream dams. To account for spatial inaccuracies between the 1:500 000 rivers and the 1:50 000 dams, the dams were buffered by 50 m. Any buffered dam that intersected a river was then assumed to be an instream dam.

A single set of priority estuaries for South Africa was derived from three estuarine systematic conservation plans (Turpie, 2005; Turpie and Clark, 2007; Rivers-Moore et al., in review) that together covered the entire coastline of South Africa. Results from Turpie and Clark (2007) were used in instances where the planning domains of the former two studies overlapped. Turpie and Clark (2007) used ten scenarios based on variation in cost and expert input – this study used scenario B5 which considers the full benefits and costs of estuaries, as well as expert input and review. Using this single set of priority estuaries, intact rivers attached to priority estuaries were identified. Any priority estuary attached to a river at a finer scale than the 1:500 000 rivers GIS layer was not included.

River reaches that could be fully incorporated into protected areas with only minor expansion were considered strategic opportunities to be investigated in terms of expanding existing protected areas. The focus here is on maintaining processes that depend on connectivity of river systems within protected areas. In many instances, processes requiring connectivity can withstand a moderate river integrity, e.g. reaches required as fish migration corridors (Chapter 6). A less stringent rule for river integrity was therefore applied by also considering moderately degraded main river systems and all tributaries regardless of their intactness. The national protected areas GIS layer was used, which includes three protected area categories – Type 1 to 3 (Driver et al., 2005; Reyers et al., 2007). Formal protected areas were defined as all Type 1 protected areas as well as Mountain Catchment Areas (legislation governing other Type 2 and 3

protected areas is less certain and these were therefore not considered). River reaches that were already fully incorporated into formal protected areas were excluded from these analyses. To qualify further under this objective, the proportion of each river reach within (1) formal protected areas, and (2) within a 2 km distance of formal protected areas was calculated. The buffer caters for river reaches falling on the boundary, or in the close vicinity, of protected areas. River reaches qualified if the proportion within (1) or (2) was $\geq 50\%$ or $\geq 75\%$ respectively.

Deriving focus areas

The MARXAN conservation planning algorithm was used as a means of integrating the multiple objectives of this study (Table 4.1) in a spatially efficient manner (Ball and Possingham, 2000; Possingham et al., 2000), together with an interface CLUZ (Smith, 2005) to view and process the data in Arcview (Version 3.2, ESRI, Redlands, CA). MARXAN helps achieve spatial efficiency through applying the concept of complementarity – areas are selected to complement each other in the biodiversity features they contain. Six steps were applied to identify focus areas:

Step 1: Set quantitative conservation targets for representation of biodiversity pattern

For river types, a conservation target of 20 % of the total length of each river type was used. For fish, the conservation target was to incorporate at least one occurrence of each endemic fish species in protected areas. In addition, only river types and fish records considered “viable” were able to contribute to achievement of conservation targets (Step 2).

Step 2: Select only “viable” river types and fish species records

The extent of “viable” river types and presence/absence of “viable” fish populations within each sub-catchment, or planning unit, was quantified and loaded into MARXAN. Here, “viable” was used broadly to refer to river types and fish populations that are most likely to persist over time. Only river types in intact systems were able to contribute to conservation targets. In addition, the total length of a river type within its sub-catchment

needed to be above a certain threshold for that sub-catchment to be considered “viable” for contributing to conservation targets for that specific river type. This length threshold differed depending on the geomorphological zone: intact mountain stream river types whose length per sub-catchment was ≥ 300 m were considered “viable”; whilst river types associated with all other geomorphological zones needed to be at least ≥ 500 m within each sub-catchment. Conservation targets for 55 river types (almost 25 %) could not be fully achieved in intact rivers. For these river types, all remaining intact river segments were selected. However, full representation through restoring rivers to an intact condition was not considered further in this study. The feasibility of restoring rivers associated with these river types should be seen as a priority for investigation.

To improve the likelihood of only selecting “viable” fish species populations, only sub-catchments containing at least 5 km of intact river length were considered suitable for achieving fish conservation targets. Four endemic fish species could not meet their conservation target in intact river segments ≥ 5 km. For these species, choices were few enough, and expert knowledge sound enough, to add in populations from the sub-catchments representing the next best options. These options were selected either from main rivers which regional experts knew could be feasibly restored, or from tributaries that had the highest percentage natural land cover modelled from the integrity assessment.

Step 3: Assess current protected areas systems and flag rivers already in protected areas as “Conserved”

Current protection levels were assessed by examining the contribution made to conservation targets by river types and endemic fish currently within formal protected areas. Based on categories from Chapter 3, well protected river types were defined as those with ≥ 100 % of their conservation target conserved in protected areas; similarly, moderately protected ecosystems, poorly protected, and hardly protected river types have at least 50 %, 5 % and ≥ 0 % of their target conserved, respectively. To assess protection levels of endemic fish within protected areas, point localities that were within formal protected areas and considered “viable” (Step 2) were investigated. Fish species

were described either as protected or not protected, depending on whether or not such a point locality existed for that species.

The contribution made to conservation targets by existing formal protected areas was acknowledged for both river types and fish species in the MARXAN analyses. This was achieved by flagging all “viable” river types and fish populations inside formal protected areas as “Conserved” before beginning the MARXAN runs.

Step 4: “Earmark” river reaches required for biodiversity processes

Earmarking planning units is a means of forcing their selection in the final MARXAN output. Sub-catchments considered of strategic importance for biodiversity processes were flagged as “Earmarked” prior to the MARXAN runs. These included sub-catchments containing free-flowing rivers, river reaches that could be fully incorporated into a protected area with only minor expansion, or intact rivers linked to priority estuaries.

Step 5: Calculate a planning unit “cost”

A planning unit cost was applied to each sub-catchment in MARXAN so that where choices existed between sub-catchments with similar biodiversity features, preference would be given to sub-catchments: (1) where $\geq 10\%$ of their area is already formally protected; or (2) containing endemic fish populations and at least 5 km of river in either an intact or moderately modified state. Each sub-catchment was assigned a uniform baseline planning unit cost and then all sub-catchments qualifying under criteria (1) or (2) were discounted to less than this baseline value. These values were determined through a series of MARXAN scenarios to test sensitivity to varying the planning unit cost and associated discount.

Step 6: Explore scenarios from different MARXAN runs and recommend focus areas

Several scenarios were examined to test the sensitivity of the MARXAN outputs. The extent to which the pattern of river integrity constrains planning was examined by comparing MARXAN results based on intact river systems against those using all river systems, irrespective of their condition. Analyses were also undertaken to test the

sensitivity of the MARXAN results to (1) varying the discount applied to sub-catchments containing a high proportion of protected areas; (2) varying the discount applied to sub-catchments containing endemic fish populations; and (3) applying a discount to the planning unit cost of sub-catchments associated with river reaches required for persistence, rather than earmarking them (Step 4).

A preferred scenario was chosen on the basis of the sensitivity analyses, and the frequency that each sub-catchment was selected in each of its MARXAN runs was then used to inform the focus areas. This frequency of selection serves as an estimate of irreplaceability (Pressey et al., 1994; Ferrier et al., 2000): sub-catchments selected in every run are irreplaceable as no options exist for their replacement; whilst sub-catchments with lower irreplaceability can be replaced by other ones, and are therefore negotiable.

RESULTS

River integrity

Patterns of river integrity in South Africa (Figure 4.2) support the notion that tributaries are less impacted than main rivers (Chapter 3), with 48 % of the river length being in an intact state when tributaries and main rivers are considered, as opposed to just 30 % when considering main rivers alone (Figure 4.3). This emphasizes the importance of tributaries for conserving biodiversity. The pattern of integrity also highlights that options in South Africa for conserving river biodiversity in intact systems are limited, since only half of the 1:500 000 river length can be considered intact.

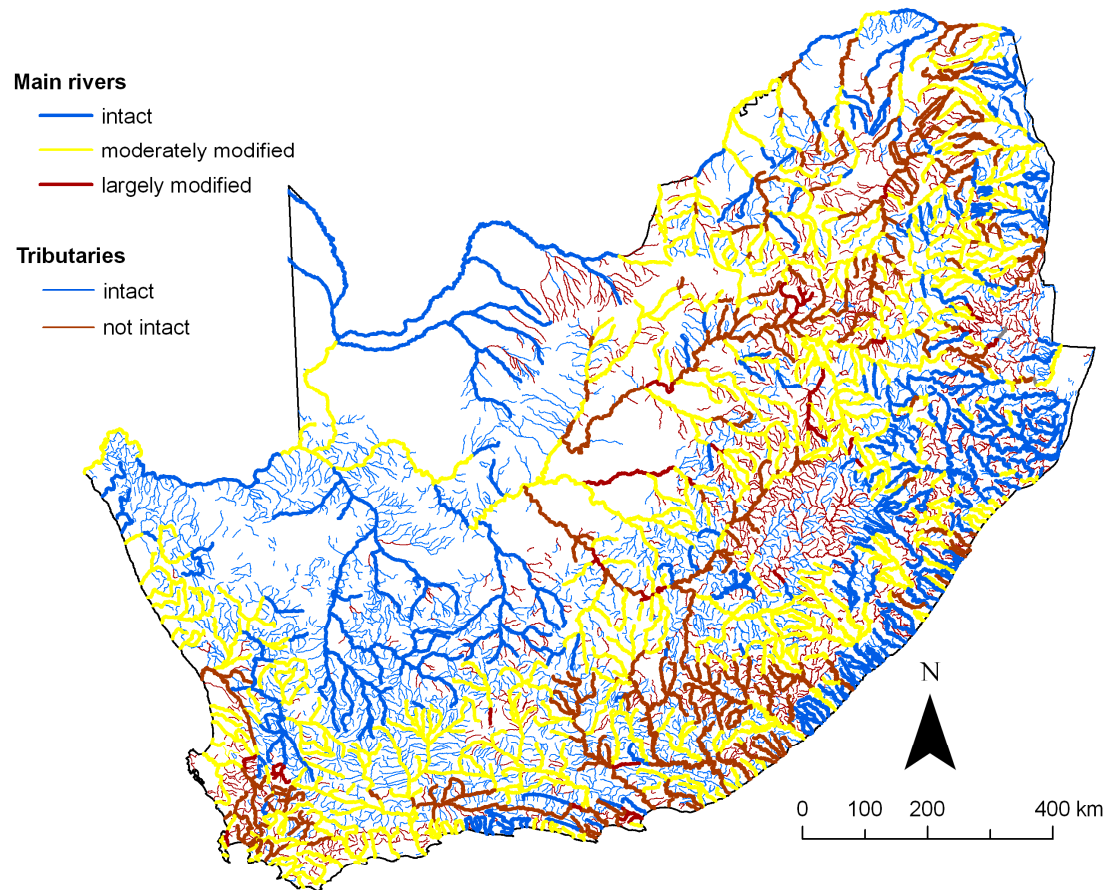


Figure 4.2 River integrity for 1:500 000 rivers of South Africa. Main river integrity is based on the present ecological state after Kleynhans (2000), while tributary integrity is based on percentage natural land cover and erosion.

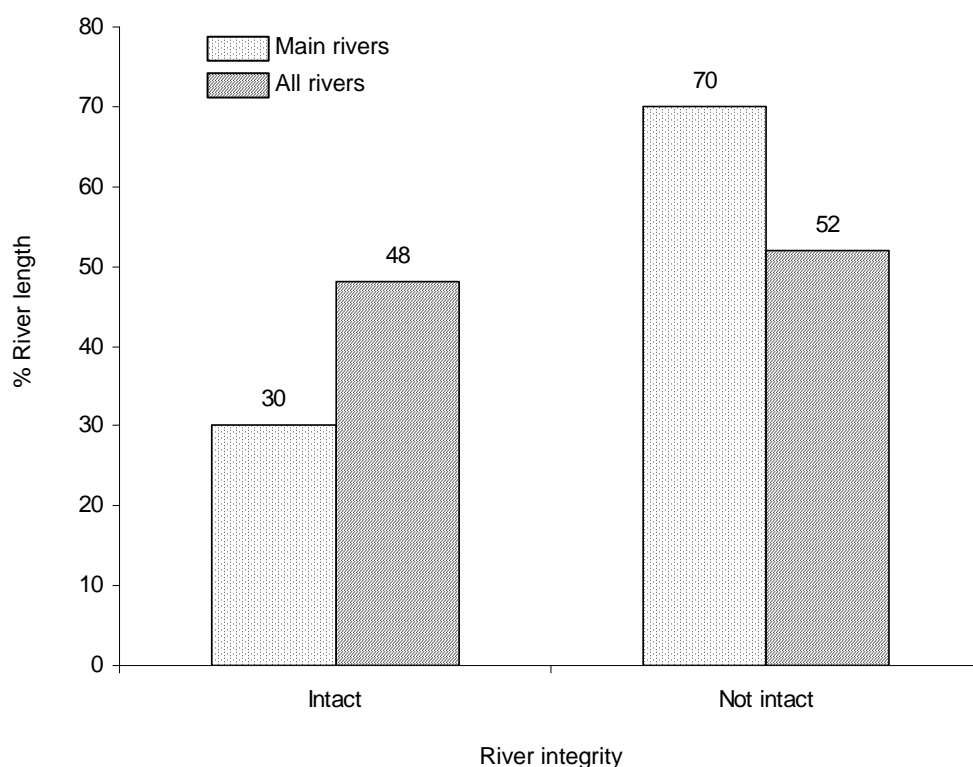


Figure 4.3 Ecological integrity of main rivers compared to main rivers and tributaries.

Biodiversity pattern and process

The combination of 30 Level 1 ecoregions, two flow variability categories and four geomorphological zones produced 222 distinct river types across the country (Figure 4.4). Over 5 300 locality records for freshwater fish endemic to South Africa were considered. These were concentrated along the permanently flowing rivers in the southern and eastern portions of the country (Figure 4.5). River systems in these areas are generally more degraded (Figure 4.2), therefore in selecting populations for achieving conservation targets, attention should be given to selecting the populations most likely to persist.

Sixty-seven free-flowing rivers were identified, distributed mainly along the eastern coast of South Africa (Figure 4.6). The largest free-flowing river reach is the White Mfolzi (424 km), followed by the Mkomazi (300 km) and Doring (280 km). Only 15 (22 %) of these are more than 100 km in length, with the majority (46 %) between 50-100 km, and the remaining 32 % comprising shorter, coastal rivers.

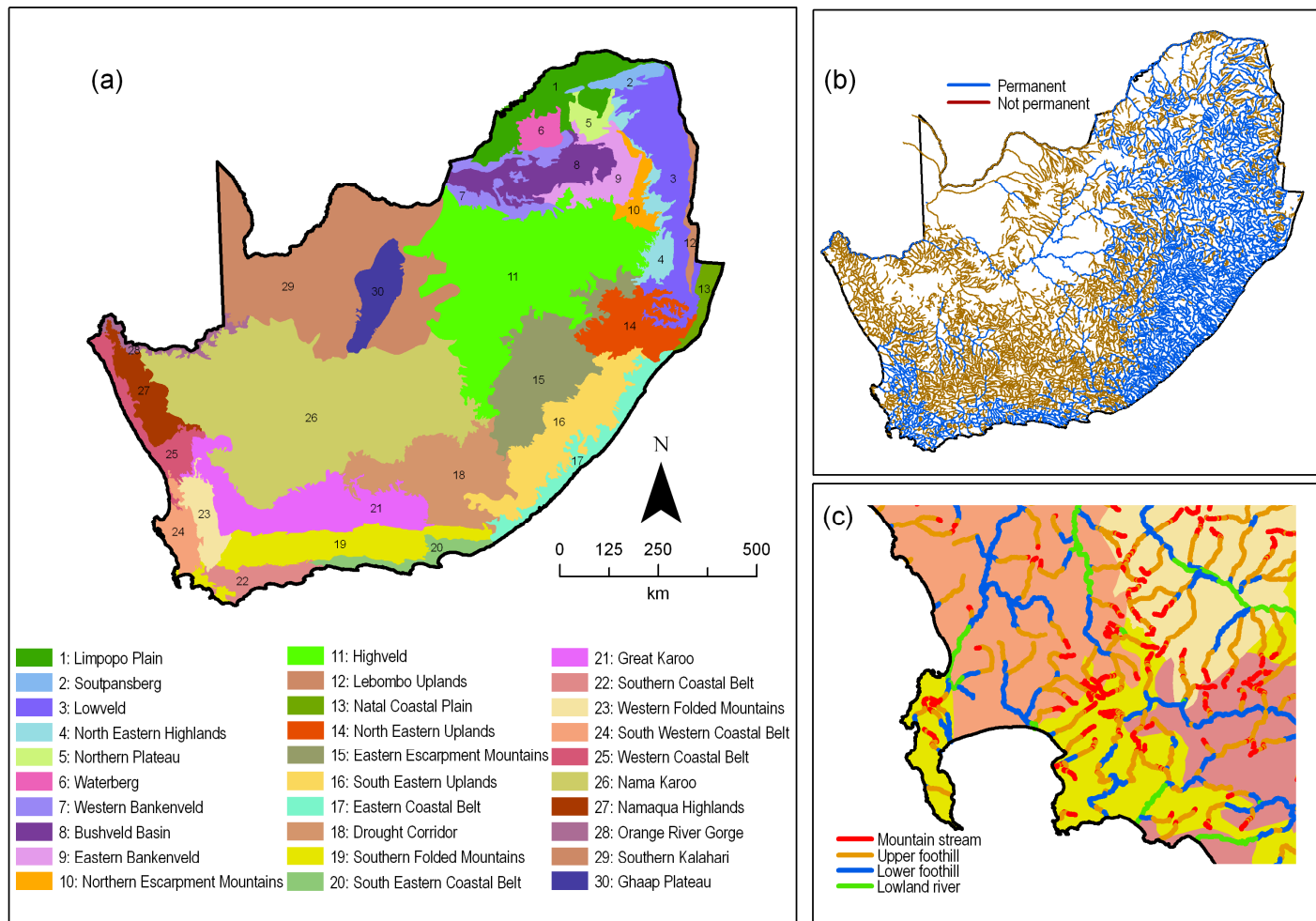


Figure 4.4 GIS layers combined to derive river types. (a) Level 1 ecoregions and (b) flow variability are shown at the country-wide scale; while (c) geomorphological zones are depicted at a finer scale for ease of viewing. Data are described in Kleynhans et al. (2005), Department of Land Affairs: Chief Directorate of Surveys & Mapping (2005) and Rowntree and Wadeson (1999) respectively.

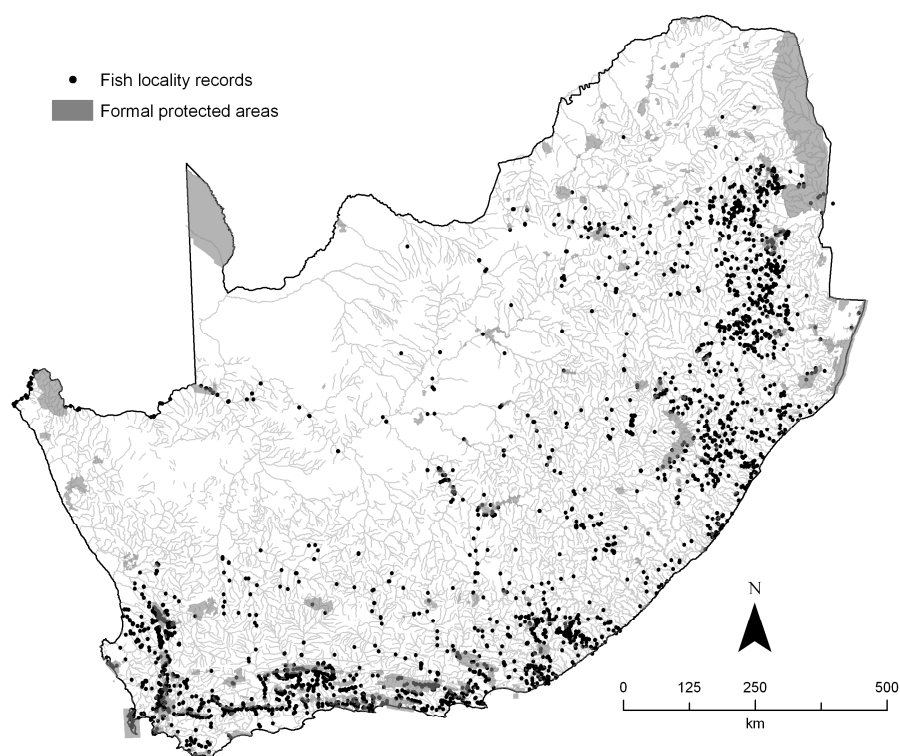


Figure 4.5 Field records of freshwater fish endemic to South Africa.

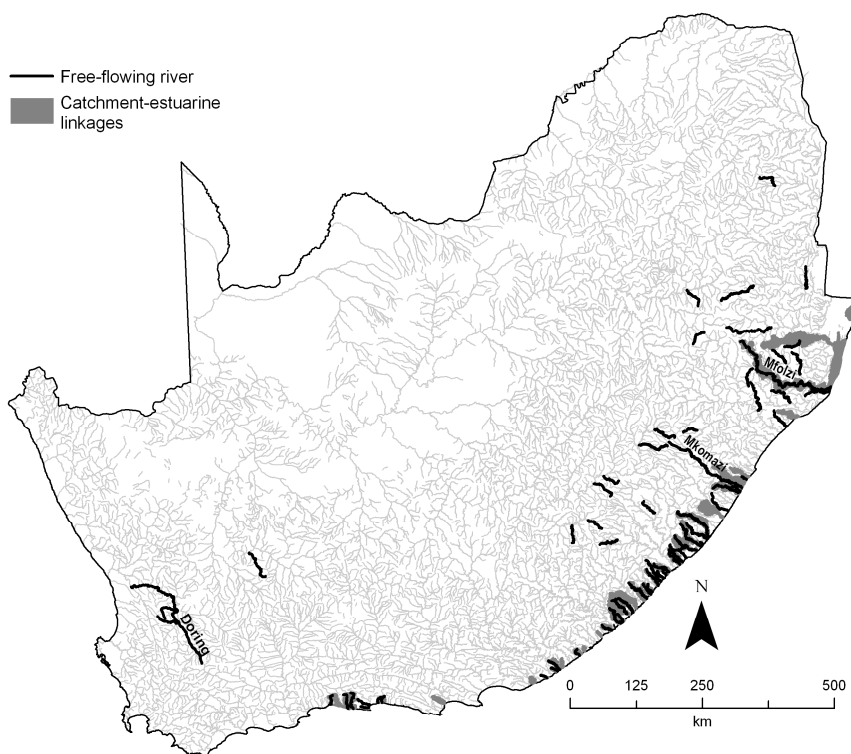


Figure 4.6 Free flowing rivers (actual river reach depicted) and sub-catchments containing intact river systems linked to priority estuaries. River names referred to in text are also given.

Almost 70 % of the 259 estuaries in South Africa are considered a priority for some form of conservation action. Only 46 of these priority estuaries (18 %) are linked to intact 1:500 000 rivers, many of which overlap with free-flowing rivers (Figure 4.6).

Protected areas that with just minor expansion could incorporate whole river reaches cluster mainly around the southern and western Cape (Figure 4.7), where there are numerous smaller protected areas in the vicinity of larger-sized Wilderness Areas or Mountain Catchment Areas. Other notable river systems are associated with larger-sized flagship protected areas, such as Kruger National Park and Greater St. Lucia Wetland Park (Figure 4.7).

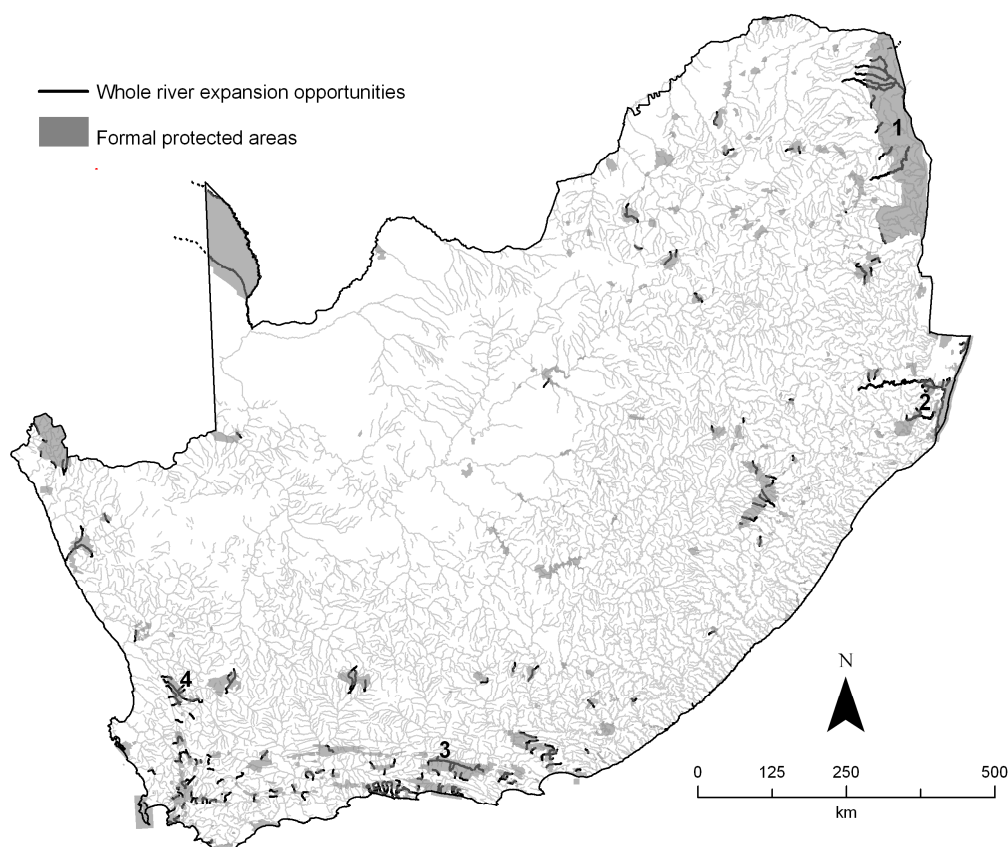


Figure 4.7 River reaches that could be fully incorporated into a protected area with only minor expansion. Formal protected areas in South Africa are also shown. Numbers 1 to 4 indicate areas referred to in text, showing Kruger National Park, Greater St. Lucia Wetland Park, Baviaanskloof Wilderness Area and Cederberg Wilderness Area, respectively.

Focus areas for expanding protected area systems

Only 21 % of the river types are moderately to well protected in the current protected area system, and more than a third are not protected at all (Table 4.2). Disaggregating these results to geomorphological zones reveals that mountain streams have the highest proportion of moderately to well protected river types, while lowland rivers have the highest proportion of river types not protected. At an ecoregion level, gaps in protection levels for river types are particularly prevalent in the arid interior and eastern coastline of the country (Figure 4.8). On the positive side, the current protected area system conserves at least one “viable” population of each freshwater fish species endemic to South Africa, and several of these species (31 of them) are captured more than once.

A total of 8 548 modelled sub-catchments were used as planning units, averaging 170 km² in size. Using a planning unit cost as a means of favouring selection of sub-catchments important for processes was ineffective, as many of the sub-catchments were not needed for representation, or their biodiversity content could be captured elsewhere in a more spatially efficient configuration. Forcing selection of these sub-catchments through earmarking them prior to beginning the MARXAN runs ensured both their selection and the maintenance of reach connectivity. Irreplaceability for scenarios that used intact rivers only was found to be insensitive to planning unit cost: even large discounts applied to sub-catchments containing either a high proportion of protected areas or “viable” endemic fish populations were ineffectual.

The preferred MARXAN scenario is shown in Figure 4.9a, in which a 90 % discount was applied to the planning unit cost of qualifying sub-catchments (see Step 5 of methods). The pattern of irreplaceability from this scenario shows that options are limited for conserving representative examples of rivers associated with the Highveld, Drought corridor, South Eastern Uplands and Eastern Coastal Belt ecoregions (Figures 4.4a and 4.9). These are the ecoregions associated with high human populations and resource use pressures. Options still exist for locating protected areas in the under-protected semi-arid ecoregions of the Nama Karoo, and to a lesser extent, Southern Kalahari and Ghaap Plateau. The pattern of irreplaceability for the MARXAN scenario that used all rivers to contribute to conservation targets (regardless of their integrity) differs markedly to the preferred scenario (Figure 4.9).

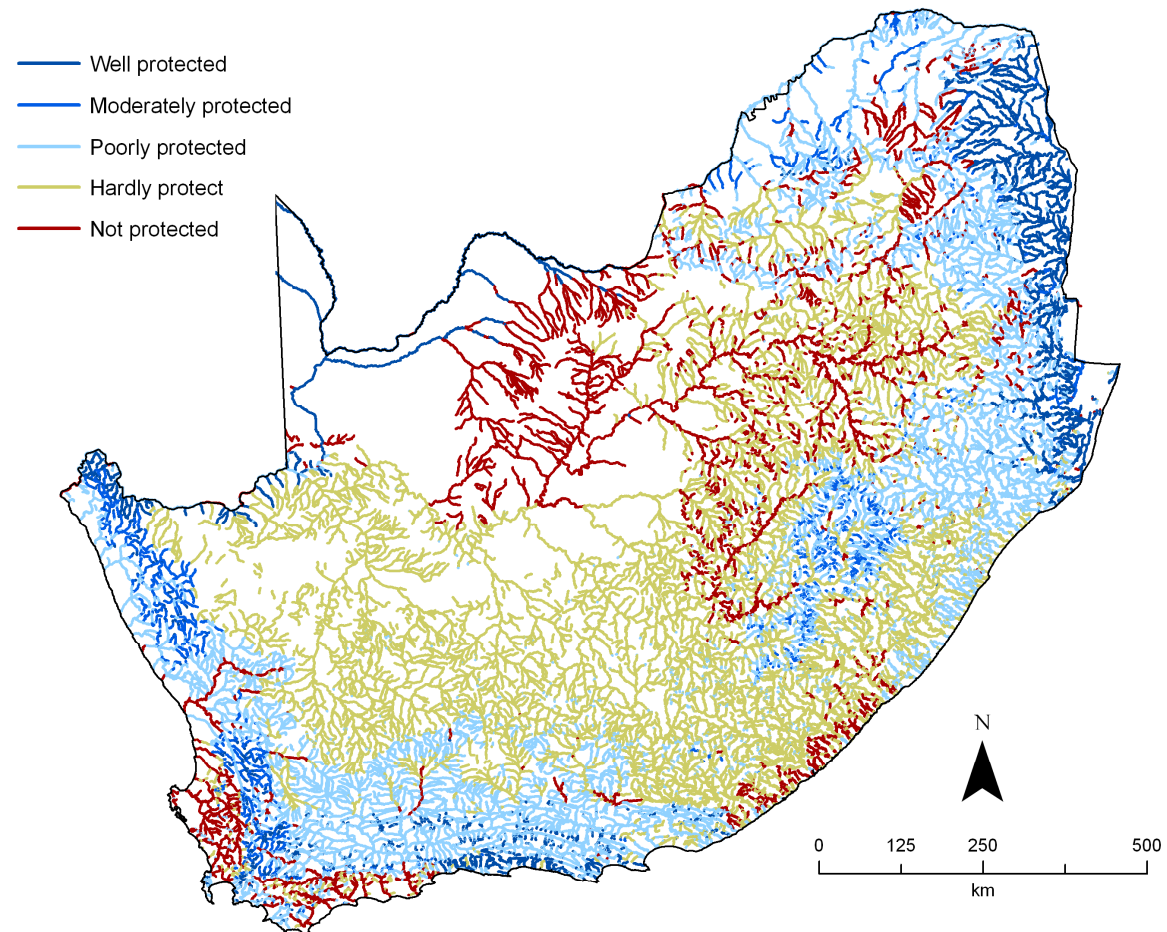


Figure 4.8 Protection levels of each river type, where well-protected, moderately protected, poorly protected, and hardly protected river types have at least 100 %, 50 %, 5 % and > 0 % of their target conserved in protected areas.

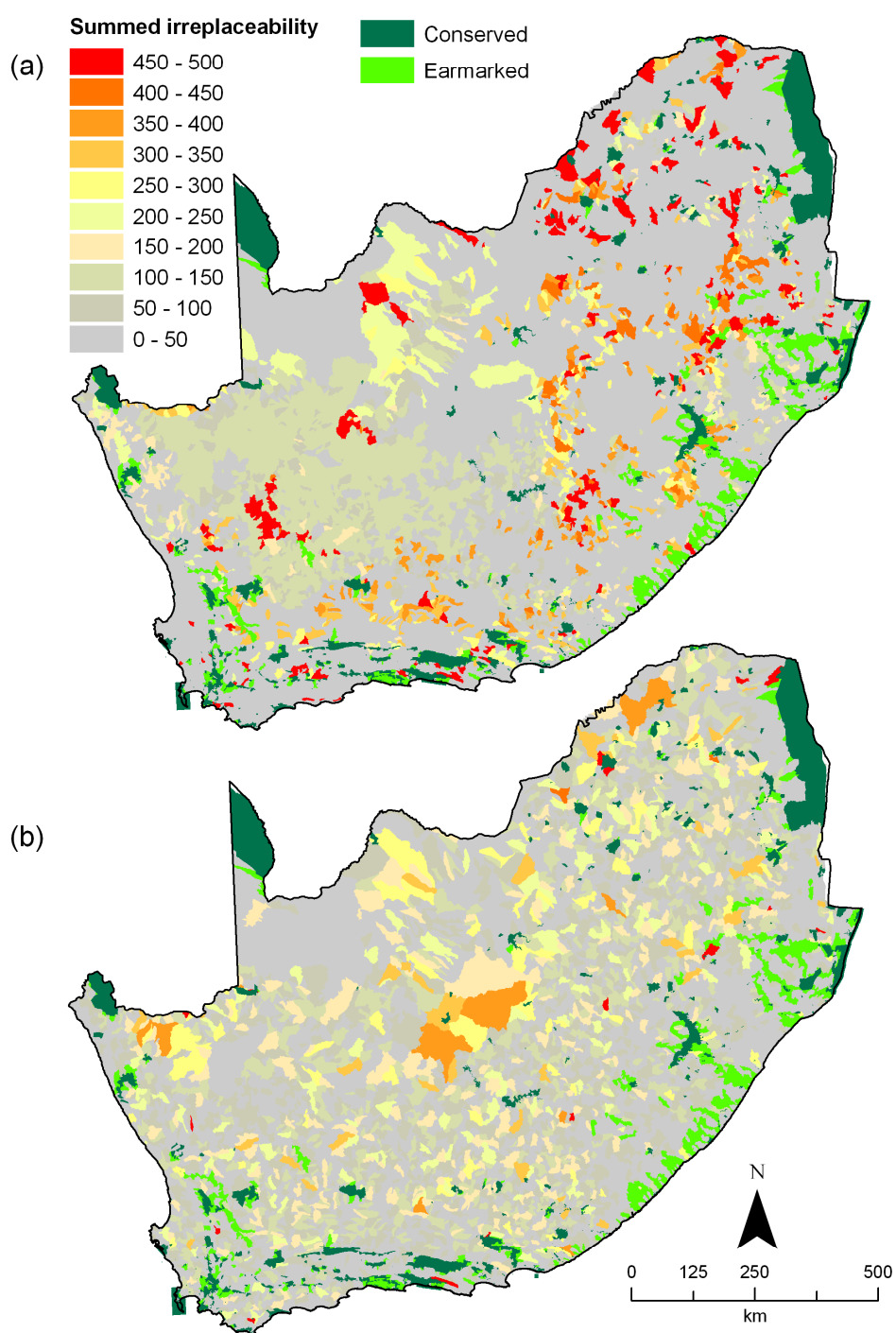


Figure 4.9 Outputs from MARXAN for (a) the preferred scenario used as the focus areas for expanding protected area systems and (b) the scenario that considered all rivers regardless of their integrity.

Table 4.2 Current protection levels for river types. Total number of river types within each protection level category are shown, as well as per geomorphological zone. Well-protected, moderately protected, poorly protected, and hardly protected river types have at least 100 %, 50 %, 5 % and > 0 % of their target conserved in protected areas.

Geomorphological zone	Not protected	Hardly protected	Poorly protected	Moderately protected	Well protected
Lowland river	30	4	6	3	6
Lower foothills	18	14	17	2	6
Upper foothills	16	13	24	3	4
Mountain streams	16	2	17	11	10
Total river types	80	33	64	19	26

DISCUSSION

This study shows, for the first time, how a conservation planning algorithm can be applied in a freshwater setting to integrate a range of multiple conservation objectives (Table 4.1). The freshwater focus areas thus identified (Table 4.3) should be investigated further at a finer scale in terms of feasibility for incorporation into South Africa's protected area system, examining issues such as potential to support persistence of freshwater biodiversity features, alignment with terrestrial and marine conservation priorities, land tenure, institutional capacity and other socio-economic constraints and opportunities (Knight and Cowling, 2007).

It should be emphasized that this study is directed specifically at expanding protected area systems. Formal protection is only one conservation mechanism, which needs to be augmented with other less restrictive mechanisms to effectively conserve freshwater biodiversity (Chapter 6; Abell et al., 2007). In addition, the focus of this study only on river ecosystems reflects a bias in the data available for freshwater ecosystems: national-scale wetland and groundwater data were not comprehensive enough for inclusion within the timeframe of this assessment. This, along with recommendations made for future improvements, should be addressed in future updates of the strategy. The important thing here is that iterative improvement over time is made

possible since this study is nested within a user-inspired, ongoing process of protected area expansion and planning coordinated by DEAT.

Table 4.3 Examples of focus areas for expanding protected area systems that would incorporate a mixture of protection strategies.

Strategy	Focus areas
Target under-protected and highly irreplaceable areas	<ul style="list-style-type: none"> • Sub-catchments with high irreplaceability in the Highveld ecoregion (Figure 4.4a), which also faces ongoing degradation (Driver et al., 2005). • Sub-catchments with high irreplaceability in the South Eastern Uplands and Eastern Coastal Belt ecoregions (Figure 4.4a), particularly those which overlap with areas identified as important for representing natural examples of large-scale catchment processes.
Target under-protected areas, where several options exist for designation of large protected areas that combine terrestrial, freshwater and marine biodiversity	<ul style="list-style-type: none"> • Sub-catchments in the Nama Karoo ecoregion which have a irreplaceability score above 50 (Figure 4.9a), where opportunities exist to align with terrestrial and marine conservation, and other socio-economic constraints in the region. An initiative similar to those of the Greater Cederberg, Baviaanskloof and Gourtiz mega-reserves initiatives should be investigated.
Incorporate natural large-scale catchment processes	<ul style="list-style-type: none"> • The relatively short coastal rivers of KwaZulu-Natal and the Wild Coast in Eastern Cape offer important opportunities for incorporating prime reference examples of systems where riverine and estuarine processes are still largely natural (Figure 4.6).

Below, generic recommendations are made to guide spatial planning for expansion of protected area systems across freshwater, terrestrial and marine realms. These recommendations are particularly pertinent to improving the way in which protected areas on land are located, designed and managed for both terrestrial and freshwater biodiversity.

Use an appropriate spatial scale and planning units

This national scale study will ultimately inform local decision making around where best to locate individual protected areas. Planning units for rivers therefore needed to be

small enough to ensure that focus areas direct protected area planning to specific places, while still considering the longitudinal and lateral linkages of freshwater systems (Chapter 5). Commonly used terrestrial conservation planning units such as grid cells, hexagons or land ownership boundaries are inappropriate for freshwater conservation planning as they do not recognize these linkages. Although use of whole catchments as planning units would fully incorporate longitudinal and lateral linkages, their use in such a study would be limited since protected areas are very seldom designated to whole catchments. From several recent studies in freshwater conservation planning, it would seem that a pragmatic solution is to split whole catchments into sub-catchments of approximately 100-200 km² in size based on river segments (Figure 4.1; Chapter 6; Linke et al., 2007; Rivers-Moore et al., in review). These sub-catchments only partially consider connectivity, and if chosen for protected area expansion will need to be augmented with other conservation mechanisms that manage external threats in connected systems to ensure that biodiversity within that protected area persists (Chapter 6).

Represent both biodiversity pattern and process

Most conservation planning efforts have focused only on representing biodiversity pattern, while fewer have specifically targeted representation of important biodiversity processes (Pressey et al., 2007). This study incorporated aspects of both. From a technical GIS perspective, this was made possible by distinguishing between a river segment and river reach – the former was used for representation of pattern; the latter to incorporate biodiversity processes.

The first objective of this study (Table 4.1) tackled representation of biodiversity pattern through setting targets for river types and endemic freshwater fish species. This objective also affords specific attention to threatened river types, defined in Chapter 3 on the basis of the proportion of the total length of each river type still intact. By definition, sub-catchments containing threatened ecosystems will have limited options for conservation in intact systems; thus consideration of threatened ecosystems is incorporated in these analyses through considering focus areas with high irreplaceability values.

In addition to representation of biodiversity pattern, objectives 3 and 4 (Table 4.1) dealt with representing examples of large-scale biodiversity processes associated with free-flowing rivers and catchment-estuarine linkages. Such opportunities are rapidly disappearing owing to the widespread and escalating degradation of freshwater systems in South Africa (Chapter 3) and worldwide (Nilsson et al., 2005; Dudgeon et al., 2006; Poff et al., 2007). These opportunities should therefore be high on the conservation agenda of all countries, and options for locating at least some of these river reaches within protected areas needs to be considered. Conserving these sub-catchments will require exploring a range of conservation mechanisms, since such vast areas are seldom isolated from human populations. In South Africa, formal protected areas can still be designated under the Protected Area Act (Act 57 of 2003) by entering a contract with communities or individual land owners – the feasibility of such agreements should be investigated.

Objectives 2 and 5 were aimed at supporting the persistence of freshwater biodiversity within selected areas. Objective 2 applies to all focus areas, using river integrity as a broad indicator of the likelihood that a river will support viable examples of biodiversity in the long term. The latter deals with connectivity of selected focus areas only, identifying strategic opportunities for incorporating whole river reaches into existing protected areas. The clustering of these strategic opportunities in the southern and western Cape illustrates the positive role of large, strategically-placed protected areas for river conservation. These areas can serve as focus areas that catalyze other formal and informal mechanisms of conservation in connected areas (Terborgh and Soulé, 1999). The mega-reserve initiatives of the Cape Region are ideal examples of this, namely the Greater Cederberg Biodiversity Corridor (<http://www.cederbergcorridor.org.za/>), and the Baviaanskloof and Gouritz mega-reserves (<http://www.baviaanskloof.net/>).

Improve planning and management of individual protected areas

The focus areas give a national indication of where benefits for river biodiversity can best be realized. However, persistence of river biodiversity within individual protected areas needs to be further supported by the way in which protected areas are delineated,

and how protected areas are managed. Delineation of new protected areas can support the persistence of freshwater biodiversity by avoiding the use of rivers as boundaries of protected areas, and maximizing hydrological connectivity within the protected area. If possible, protected area boundaries should strive to incorporate the full range of geomorphological zones within each ecoregion and flow category (Figure 4.4); if captured on the same river system, this will not only improve representation of river types, but will also incorporate river connectivity.

A first step towards enhancing management effectiveness of freshwater biodiversity within protected areas is to ensure that protected area management plans explicitly address freshwater conservation issues. These include addressing issues within the protected area (e.g. ensuring that tourist lodges and roads have minimal impact on river systems), as well as processes and threats external to the boundaries of the protected area.

Use irreplaceability and protection levels to inform focus areas

The pattern of irreplaceability used to guide freshwater focus areas (Figure 4.9a) is not a minimum set of sub-catchments required to achieve conservation targets. This study did not develop a single-set solution to depict focus areas because these do not provide an indication of whether a selected sub-catchment is essential for achieving conservation targets or whether it can be replaced by other ones and is therefore negotiable. Understanding which areas are negotiable is important for integrating this assessment into the overarching protected area expansion strategy, which considers a whole multitude of other ecological, social and economic objectives.

However, it is critical that this irreplaceability map is interpreted correctly within the context of protected area expansion strategy. Selecting focus areas only from sub-catchments of high to moderate irreplaceability will undermine representation, since some low irreplaceability sub-catchments will still be needed to achieve conservation targets. This is particularly relevant for ecoregions where both irreplaceability and protection levels are low – locating at least one protected area in these ecoregions should be regarded as a conservation priority. In these instances, there will be a number of options from which to choose and location of the protected area should be further

guided by other strategic objectives, such as terrestrial conservation priorities or socio-economic constraints.

Exploring the sensitivity of the irreplaceability results to varying planning unit costs was also informative. Insensitivity of the irreplaceability outputs to large discounts applied to planning unit costs of sub-catchments containing protected areas suggest that it is spatially inefficient to improve representation by expanding existing protected areas. New protected areas that take cognizance of persistence issues should rather be created in under-protected areas.

Choose focus areas that incorporate a mixture of protection strategies

Figure 4.9a shows focus areas that would achieve a range of different objectives. First, earmarked areas highlight opportunities for improving persistence of river systems already in protected areas, or for representing key biodiversity processes. Earmarked areas have been selected at the scale of a river reach (Figure 4.1) and ideally the entire reach needs to be included. Second, sub-catchments with a high irreplaceability value have very few substitute areas for meeting conservation targets. Protecting rivers in these sub-catchments will target river types or fish species that have very limited distributional ranges in South Africa, either naturally or because these are the last remaining “viable” examples. Third, as irreplaceability decreases, options for protected area placement increase. In these areas, protected area designation should be guided by other strategic objectives of the overarching protected area expansion strategy. Finally, areas of little benefit for protected area expansion (e.g. irreplaceability 0-50 on Figure 4.9a) should be avoided.

A common approach to prioritizing conservation action is to combine irreplaceability with vulnerability – a measure of the future risk of degradation (Margules and Pressey, 2000). The notion here is that areas of high irreplaceability and high vulnerability should be secured before those associated with lower vulnerability. Linke et al. (2007) use a similar approach for exploring management options for river conservation planning. This framework is useful for planning that considers a range of conservation mechanisms; however, its use is limited in the context of protected area planning. Areas of high irreplaceability and high vulnerability are often areas where

land-use conflict and land purchase costs are high – conserving ecosystems in such situations is often more pragmatically achieved through mechanisms other than formal protected areas. On the other hand, areas of low vulnerability that are currently under-protected often offer more cost-effective opportunities for the designation of large protected areas while still improving representation. It is therefore recommended that protected area expansion strategies incorporate a combination of strategies in their schedule of action (Table 4.3), balancing protection strategies that focus on rescuing threatened biodiversity with strategies that prevent the biodiversity that is currently secure from becoming threatened.

Embed planning into an ongoing and adaptive implementation process

This study is embedded in a real-world iterative process of protected area planning by South Africa's government department responsible for protected area planning and management (DEAT). The overall strategy that this study informs updates an outdated protected area strategy, which did not include freshwater biodiversity considerations or systematic conservation planning principles, and will itself be updated and refined in the future. To support the process of adaptive improvement, the scope of this spatial assessment needs to be extended, and several limitations will need to be addressed.

First, freshwater ecosystems other than rivers need to be considered. This will require addressing data gaps for wetlands and groundwater at an appropriate scale for country-wide systematic conservation planning. It will also require identifying a sub-set of estuarine focus areas for protected area expansion, from the numerous priority estuaries already identified as requiring some form of conservation. Second, transboundary river basins along South Africa's northern boundary should be included, focussing specifically on improving the persistence of river biodiversity associated with South Africa's existing protected areas. Finally, almost 25 % of the river types cannot achieve their targets in intact river systems. Restoration options for these river types should be strongly considered, but owing to the complexity of such analyses, were not considered here. This influences the final pattern of irreplaceability used to inform focus areas (Figure 4.9a). For example, the reason that the south-western portion of the

country is depicted of limited value for protected area expansion is because there are no intact river systems remaining.

The results indicate that this spatial assessment is strongly dependent on river integrity. However, the data used for main river integrity (Kleyhans, 2000) needs updating, and the level of confidence in the modelled tributary data is unknown. In addition, the land cover data used for modelling tributary integrity (Fairbanks et al., 2000) is out of date and underestimates the extent of land degradation (Thompson et al., 2008). Improving the confidence of the river integrity data would greatly support the credibility of the final product.

Free-flowing rivers identified in this study serve as an initial basis around which regional experts can debate. Some of these rivers may not qualify as free-flowing owing to limitations of the input data: (1) farm dams built after 2005 have not been included in the connectivity analyses; (2) weir data were not included as there is no such national GIS layer; and (3) water transfer schemes were not explicitly included in the analyses (however, for main rivers they were accommodated implicitly in the assessment of river integrity). The buffering technique used may also disqualify some rivers which are indeed free-flowing since off-stream dams within 50 m of a river will be classified as instream dams.

The issue of integrating freshwater, terrestrial and marine spatial plans has received very little attention worldwide. This is a key area of research that needs to be addressed in the next iteration of this study. While it is intuitively appealing to run a single MARXAN analysis for both terrestrial and freshwater biodiversity to derive a fully integrated pattern of irreplaceability, this can result in a loss of realm-specific information. For example, terrestrial planning units used to identify focus areas are orders of magnitude smaller than freshwater sub-catchments – 0.01 km² in size (S. Holness, unpublished data) compared to the average size of 170 km² for sub-catchments. Combining the assessment at the level of a sub-catchment would therefore result in a loss of terrestrial-specific detail. Consequently, alternative methods of integration also need to be explored.

CONCLUSIONS

The development of approaches to protected area planning for freshwaters is a timely topic given the ongoing degradation and massive threats faced by these ecosystems (Revenga et al., 2005; Dudgeon et al., 2006), and the subsequent surge of recent calls for urgent attention to be given to protecting freshwater biodiversity (Abell, 2002; Dunn, 2003; Fitzsimons and Robertson, 2005; Abell et al., 2007). This analysis has been specifically designed for guiding expansion of formal protected area systems. Realistically, protected areas can only play a partial role in overall efforts to conserve freshwater biodiversity, and will need to be supplemented with other less stringent conservation mechanisms. These could include, for example, managing threats in systems that are connected to those within protected areas, incorporating restoration options for river types that cannot meet their conservation targets in intact systems, conserving a broader range of freshwater species, increasing the species conservation targets so as to build in a suitable level of redundancy and resilience, and including representation and persistence of wetlands.

This assessment suggests that large wilderness areas delineated according to sub-catchment boundaries have huge potential for representing natural examples of both freshwater biodiversity pattern and processes. Whatever their size, protected areas have the powerful ability to catalyze conservation activities in the surrounding catchments, providing the stimulus for the implementation of effective integrated catchment management. Protected area managers can learn from recent management practices in the Kruger National Park, South Africa (O’Keeffe and Rogers, 2003; Pollard et al., 2003), where explicit consideration of freshwater issues beyond the Park’s boundary are now an intimate part of their adaptive management strategy, working towards inspiring surrounding communities and fostering a spirit of cooperation for conserving freshwater ecosystems both within and outside protected areas.

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CHAPTER 5.
A FRAMEWORK FOR INCORPORATING
PERSISTENCE INTO THE DESIGN OF
FRESHWATER CONSERVATION AREA
NETWORKS

University of Cape Town

A FRAMEWORK FOR INCORPORATING PERSISTENCE INTO THE DESIGN OF FRESHWATER CONSERVATION AREA NETWORKS

ABSTRACT

This paper presents a framework for planning for the persistence of freshwater biodiversity. A distinction is made between the complementary procedures of planning for persistence and planning for retention. The former focuses on maintenance of natural processes that support biodiversity and allow it to evolve naturally over time, while the latter deals with minimizing the impacts of threatening anthropogenic processes. The persistence framework is based on four principles originating from concepts developed in freshwater ecology and terrestrial conservation planning: selecting ecosystems of high ecological integrity; incorporating connectivity; incorporating populations that have a high probability of persisting; and identifying additional fixed spatial components of processes (e.g. high water yield areas). Planning for persistence is severely limited by our understanding of natural processes. Several surrogate measures are suggested to address such knowledge deficiencies. These surrogates of persistence are essentially hypotheses that should be tested over time. Broad recommendations are provided on how to incorporate each of the principles into the conservation planning process to ensure spatial efficiency. From an implementation perspective, this study recommends allocating different levels of protection to areas depending on the specific function of the area and its sensitivity to human impacts. The importance of embedding conservation plans into integrated water resources management is also emphasized.

KEY WORDS: Systematic conservation planning; river; connectivity; complementarity; biodiversity; integrated water resources management

INTRODUCTION

Despite two decades of research and development in terrestrial settings, systematic conservation planning has only recently found its way into planning for the conservation of freshwater ecosystems and their associated biota (Chapter 2). Systematic conservation planning aspires to achieve representation and persistence of biodiversity in a resource efficient manner (Margules and Pressey, 2000). Representation refers to the need to adequately conserve the full variety of biodiversity features in a planning region, while persistence is concerned with ensuring that biodiversity within the region persists and evolves naturally over time. Efficiency seeks to achieve representation and persistence within the context of financial, social and economic constraints. The majority of systematic conservation planning efforts (over 80 %) have focussed only on efficiently representing biodiversity pattern – those biodiversity features, such as species and ecosystem types, that are mapped and regarded as static (Pressey et al., 2007). They have done less well at incorporating the spatial and temporal aspects of persistence, such as providing connecting corridors between conservation areas for species that depend on seasonal migration (Cowling et al., 1999).

No matter what the planning realm – terrestrial, freshwater, marine or combinations of these – plans that fail to address issues of persistence will not secure biodiversity in the long term. While this shortcoming has been increasingly recognized within the context of terrestrial conservation planning (Pressey et al., 2007), it is particularly obvious in freshwater settings. In the case of all but the most isolated freshwater systems, persistence of biodiversity at any particular location is fundamentally dependent on the upstream drainage network, the surrounding land use, the riparian zone and associated wetlands, and in the case of migratory fauna – downstream reaches (Dudgeon et al., 2006). Planning for persistence in freshwater conservation therefore cannot be ignored.

The absence of both frameworks and tools for dealing with persistence in freshwater settings is one of the main reasons preventing the earlier application of systematic conservation planning in the freshwater realm. The recent incorporation of longitudinal connectivity into conservation planning algorithms is a major advance in

the development of tools for dealing with this problem (Chapter 6; Linke et al., 2007; Moilanen et al., 2008). However, conceptual frameworks that consolidate principles for incorporating persistence into freshwater conservation planning are still lacking. Several concepts that provide insights into planning for persistence have emerged over the last two decades from both freshwater ecology and terrestrial conservation planning. This paper consolidates some of these concepts in deriving four principles for incorporating persistence into the design of a freshwater conservation area network.

The principles presented here are not new to the literature, being based either on freshwater ecological theory, or concepts borrowed from terrestrial conservation planning. The novelty of this work lies in assembling these principles into a framework aimed specifically at planning for persistence of freshwater biodiversity. The chapter begins by clarifying what is meant by planning for persistence, and hence defining the scope of this study. It then outlines each of the principles for incorporating persistence into the design of freshwater conservation area networks, describing their basis and rationale. Finally, it provides guidance on how to incorporate these principles into the conservation planning process to design an efficient and pragmatic freshwater conservation area network. Chapter 6 demonstrates the application of this framework, using a case-study from South Africa.

PERSISTENCE, RETENTION AND VULNERABILITY

The persistence of biodiversity within a planning region is dependent on two aspects: maintaining the natural processes that support and generate biodiversity, such as the aforementioned seasonal migration corridors; and managing anthropogenic processes that threaten biodiversity. Over the years, several terms and approaches have been developed in systematic conservation planning for incorporating these two aspects into planning and it is useful to distinguish between these.

“Persistence” is usually applied within the context of designing conservation area networks that maintain natural processes (Cowling et al., 1999; Gaston et al., 2002). Although planning for biodiversity pattern may incidentally incorporate some natural processes, it tends to ignore those that operate over large areas, or require special spatial configurations (Pressey et al., 2007). Planning for persistence of biodiversity

therefore embeds the conservation areas selected for representation into a conservation area network – a network designed to maintain key natural processes (Margules and Sarkar, 2007). Several approaches have been used in terrestrial settings to design such conservation area networks, such as incorporating connectivity into spatial design criteria (Cowling et al., 1999; Possingham et al., 2000), considering species persistence issues (Nicholls, 1998; Rodrigues et al., 2000; Williams and Araujo, 2000; Gaston, Pressey and Margules, 2002), accommodating processes associated with “umbrella” species such as large mammals (Carroll et al., 2001; Kerley et al., 2003), and identifying “fixed spatial components of processes” (Rouget et al., 2003; Rouget et al., 2006). The latter term – fixed spatial component – refers to process surrogates that are spatially fixed and can be mapped, as opposed to spatially flexible aspects of persistence that depend on how the conservation area network is configured. Most of these concepts are assimilated into the persistence framework presented in the subsequent section.

The terms “retention” or “vulnerability” are used in the published literature when dealing with threatening anthropogenic processes (Cowling et al., 1999; Gaston, Pressey and Margules, 2002). Retention strategies seek to minimize ongoing habitat loss or degradation during the gradual process of implementing a conservation plan by strategically scheduling conservation action. Vulnerability provides an approach to achieving retention by measuring the likelihood that a site will be transformed by human activities. A scheduling framework can then be developed using site vulnerability in relation to its irreplaceability (Margules and Pressey, 2000), where irreplaceability is a measure of the likelihood that the site will be required to achieve conservation targets (no options exist for the replacement of totally irreplaceable sites; whilst sites with lower irreplaceability can be substituted with other ones). In theory, sites with a high irreplaceability and a high vulnerability will require the most urgent conservation action, since the likelihood and consequence of biodiversity loss is highest.

Retention strategies can be extended from this narrow approach of scheduling conservation action to include a broader suite of mechanisms for dealing with anthropogenic threat (Gaston et al., 2002; Pressey et al., 2007). An assessment of vulnerability and its underlying causes can also be used to inform the development of appropriate conservation mechanisms and management objectives for individual sites

within the conservation area network so as to minimize threats to biodiversity. These approaches to dealing with threatening anthropogenic processes – scheduling conservation action, assigning appropriate levels of conservation, and developing management plans – are usually accomplished at the end of designing a conservation area network. However, an assessment of vulnerability can also be applied during the conservation planning process. For example, in designing a conservation area network, areas of high vulnerability can be avoided where there are choices to go elsewhere.

Vulnerability is essentially a critical input into a retention strategy which seeks to minimize biodiversity loss in the face of ongoing and dynamic anthropogenic threats. In this study, the term “planning for persistence” is adopted when referring to maintaining natural processes that allow biodiversity to persist, and the term “planning for retention” is used to refer to anthropogenic processes that threaten the persistence biodiversity. The scope of this framework is limited to planning for persistence. However, it should be emphasized that both aspects are critically important to incorporate into freshwater conservation planning since the key drivers of decline in freshwater biodiversity are human impacts (Dudgeon et al., 2006).

PLANNING FOR PERSISTENCE IN FRESHWATER SETTINGS

This section outlines the conceptual basis of the framework, which consists of four key principles. These principles are essentially surrogates for the persistence of biodiversity. For each principle, this section explores the main rationale for its inclusion, the key processes it is likely to incorporate, insights gained from concepts in freshwater ecology or conservation planning, and how to assemble the information into a format suitable for conservation planning.

Principle 1: Select systems of high ecological integrity

In its broadest sense, ecological integrity can be defined as the undiminished ability of an ecosystem to continue its natural path of evolution, its normal transition over time, and its successional recovery from disturbances (Westra et al., 2000). A principle firmly entrenched in conservation planning is to use ecological integrity as a screening mechanism, preferentially selecting ecosystems of high ecological integrity for representation (Groves, 2003). Where this is not possible the system with best restoration potential should be selected. This principle is associated more with incidental capture of functional processes rather than targeting specific spatial requirements of natural processes. Selecting systems of high ecological integrity incidentally captures a multitude of fine-scale biological processes such as competition, predation, and small-scale disturbance and recolonization dynamics (Figure 5.1). It also captures several functional physical and chemical processes that shape the structure and functioning of freshwater systems (Figure 5.1). In freshwater ecosystems that exhibit strong ecological connectivity, capturing systems of high ecological integrity also incorporates large, landscape scale processes associated with the natural flow regime, since these systems by definition would have flow regimes operating similarly to their natural pattern of variation.

The concept of ecological integrity is well-established in freshwater ecology (Boulton, 1999), although still the subject of considerable debate (Gergel et al., 2000). Multiple indicators have been developed within the context of bioassessment programmes to monitor changes to one or more attributes of the five primary attributes of integrity (Figure 5.1), namely energy sources, physical habitat, flow regime, water quality, and biotic interactions (Karr et al., 1986; Poff et al., 1997). Directly assessing the ecological integrity of freshwater systems using these indices requires site-based field data that are generally only available for major rivers in a planning region. Data are lacking for smaller streams, which are often the last refuges for much biodiversity (Chapter 4; Freeman et al., 2007). Therefore, conservation planning is largely dependent on the use of indirect surrogate measures for mapping this ecological integrity, coupled wherever possible with existing field data and expert knowledge.

Land cover data are the most common surrogate measures used to infer information about the impact that human activities have on freshwater systems (Stein, Stein and Nix, 2002; Linke et al., 2007; Norris et al., 2007; Thieme et al., 2007). Information such as water use, pollution and sedimentation can be inferred from analysing the extent and degree of agriculture, urban areas, and degraded, eroded or impervious surfaces (Allan, 2004). An index of hydrological alteration and fragmentation can be obtained using indices such as dam and road density, or number of road-stream crossings (Abell et al., 2002). Ideally, these indices should be evaluated at multiple spatial scales to incorporate both local- and catchment-scale influences. For example, regression models examining the utility of land cover surrogates indicate that land cover in a riparian buffer zone is a significant predictor of river integrity, but land cover throughout the catchment is a more powerful predictor (Snyder et al., 2005; Amis, et al., 2007). In terms of land cover composition, percentage natural vegetation within the catchment is the primary predictor of river integrity in semi-arid planning regions dominated by irrigated agriculture (Amis et al., 2007), whilst the extent of impervious surfaces is the primary predictor in wetter planning regions where a mixture of residential development and forested land predominates (Snyder et al., 2005). These results indicate that, where no other data exist, the extent of natural vegetation or impervious surface within riparian buffers, sub-catchments and catchments can be useful surrogates.

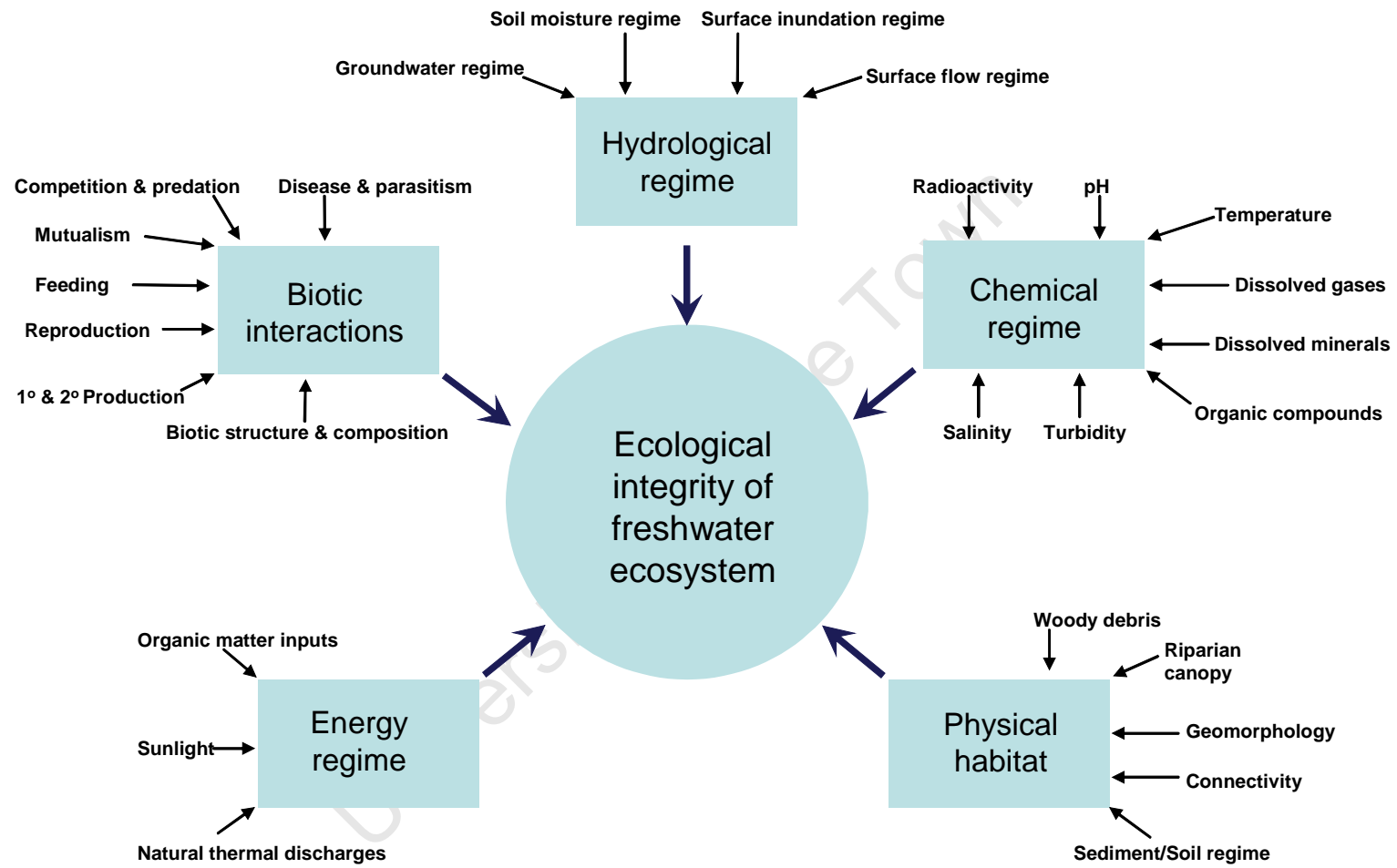


Figure 5.1 The biological, physical and chemical processes thought to be key determinants of freshwater ecological integrity (modified from Karr et al., 1986). An ecosystem is assumed to have a high ecological integrity when these determinants operate within their normal range of variation.

Principle 2: Incorporate connectivity

The persistence of most freshwater ecosystems is, directly or indirectly, maintained through connectivity along three spatial dimensions (longitudinal, lateral and vertical), and a temporal dimension linked to the availability of surface water over time, with flow regimes being of crucial importance (Ward, 1989; Poff et al., 1997; Pringle, 2001). Over the years, the inter-dependence of these dimensions has also been highlighted, emphasizing the importance of considering connectivity across all dimensions (Ward, 1989; Pringle, 2001; Ward and Tockner, 2001; Freeman et al., 2007).

Longitudinal connectivity describes the upstream-downstream continuum of rivers (Vannote et al., 1980), which supports processes such as free passage of biota, nutrients, energy, organic matter and sediment. The recovery of disturbed habitats is often dependent on maintaining longitudinal connectivity with undisturbed habitats, which serve as sources of recolonization (Frissel, 1997). Providing free passage for migration between habitats is essential to the persistence of species depending on a variety of habitats to complete their life-cycle. This is also true for species that rely on altitudinal migration to find optimal stream temperatures – a particularly important aspect to consider in the context of climate change. Maintenance of longitudinal and lateral connectivity, combined with natural flow regimes, also affects the ability of water to erode, transport, sort, and deposit alluvial materials (Dollar et al., 2007). This in turn results in a dynamic equilibrium, where the river maintains its structural pattern, functionality and hence biodiversity over time.

Lateral connectivity refers to the interactive pathways from the river channel to the surrounding catchment (Naiman and Décamps, 1997). River-floodplain interconnections are especially pronounced and strongly dependent on natural seasonal flooding, which supports processes such as passage to spawning habitat, recolonization events, transport of sediment and organic matter, and the subsequent maintenance of patch dynamics (Junk et al., 1989). In addition to floodplain wetlands, other types of wetlands play varying roles in supporting processes such as water purification, maintenance of groundwater recharge and discharge, and the regulation of flow, temperature, sediment and erosion dynamics. The maintenance of lateral linkages between the river channel and its riparian zone also captures several natural processes

such as bank stabilization, temperature regulation through shading, regulation of nutrients and sediments, and movement of freshwater-dependent biota (Naiman and Décamps, 1997; Ewel et al., 2001; Nakano et al., 1999; Nakano and Murakami, 2001).

Vertical connectivity describes exchange pathways between surface waters and groundwater (Ward, 1998), which supports processes such as the regulation of water levels during periods of reduced rainfall, water temperature and dissolved mineral content (Malard et al., 2002; Baker et al., 2003). Maintenance of refuge pools in seasonally-flowing rivers, or in times of drought, are critical to the survival of aquatic-dependent biota and are frequently dependent on groundwater. Water tables that maintain riparian vegetation are also often sustained by groundwater (Horton et al., 2001; Baird et al., 2005). Temporary storage within the hyporheic zone – the region of mixing between surface water and groundwater – can strongly influence nutrient cycling and stream metabolism by extending the period of contact between nutrients and associated biota (Sophocleous, 2002). Preliminary investigations also indicate that the hyporheic zone may be used as a refuge during floods and droughts, and as a nursery by benthic invertebrates (Boulton et al., 1998).

Incorporating longitudinal connectivity into a conservation design in its entirety is achieved by conserving whole river systems, ensuring that there are no artificial barriers impeding upstream-downstream exchange pathways and that the flow regime operates within its normal range of variation. In practice, it is difficult to find large free-flowing river systems; where these do exist they should be afforded the highest conservation priority (Chapter 4; Nilsson et al., 2005). Longitudinal connectivity for other utilized rivers should be enhanced as far as possible by management activities such as removal of small artificial barriers where feasible (O'Hanely and Tomberlin, 2005), construction of appropriate fish passages (Schilt, 2007), and the adoption of water allocation and release schemes that adhere to environmental flow requirements (Postel and Richter, 2005).

Lateral connectivity can be broadly incorporated into conservation planning through the use of catchments as planning units. This emphasizes the need to manage both the land and the smaller stream network within selected catchments. Primary drainage catchments of river systems are generally too large to provide sufficient detail

required for conservation planning, and it is usually necessary to delineate sub-catchments within these at a scale that provides an appropriate distinction between the different habitat types and species occurrences (Chapter 4; Higgins et al., 2005). Geographic Information System (GIS) tools, such as Arc Hydro (Maidment, 2002) and HydroSHEDS (Lehner et al., 2006), have been developed to aid the delineation of sub-catchments. At a finer level of detail within sub-catchments, lateral connectivity can be considered by targeting the persistence of the most important functional lateral zones (Saunders et al., 2002). For example, new technologies from airborne and satellite remote sensing platforms have recently been developed for identifying wetlands (Davidson and Finlayson, 2007; Hamilton et al., 2007) and riparian zones (Goetz, 2006), which have recently been applied in conservation planning (Ausseil et al., 2007). Where this level of information is not available, conservation plans can cater for varying levels of lateral connectivity within a sub-catchment by using rules of thumb for delineating buffer zones surrounding the freshwater ecosystems of concern (Jones et al., 2006).

Incorporating vertical connectivity into a spatial design requires identifying areas that need to be managed to prevent activities that may threaten groundwater quality and quantity. Mapping patterns of groundwater discharge and recharge at a scale useful for conservation planning can aid this process. In the absence of consistent field observations across the planning region, discharge and recharge patterns can be mapped using a combination of environmental surrogates, which may include topography, geological permeability, groundwater depth surfaces and presence of groundwater dependent vegetation (Baker et al., 2003).

Finally, the maintenance of connectivity along all spatial dimensions is strongly dependent on the temporal dynamics of the natural flow regime (Pringle, 2001). Since most rivers are utilised by people, maintaining a truly natural flow regime is often impossible. A large body of research and development exists for addressing environmental flow requirements that seek to balance the needs of people and the environment (Tharme, 2003; Dudgeon et al., 2006). Integrating this research with planning for retention holds enormous potential, but has not been accomplished to date. The explicit identification of conservation areas during planning for persistence should

be regarded as a necessary step in the process of negotiating environmental flow requirements and water allocations, and hence, integrated water resources management.

Principle 3: Incorporate populations that have a high probability of persisting

Targeting systems with a high ecological integrity and incorporating connectivity into conservation design serve as generic surrogates for the persistence of populations in a conservation area network, and will sometimes be the only ones available. As a further safe-guard, this principle explicitly addresses the persistence of species selected for representation in a conservation plan. Considerations specific to the persistence of each species population include: incorporating access to all critical habitat required over the life-time of each species; identifying areas that serve as spatial refugia and incorporating linkages between these and the populations; replication within the planning region in areas that are unlikely to be influenced by the same natural or human disturbances; and incorporating populations or metapopulations that are large enough to prevent extinction from random demographic and genetic events (Moyle and Yoshiyama, 1994).

Limited knowledge of the habitat requirements and population dynamics of most freshwater species poses a major challenge to evaluating persistence of populations. Whilst the use of genetic markers to study spatial distribution patterns of populations has helped to quantify spatial and temporal requirements (Neville et al., 2006), the techniques are expensive and only available for a limited number of species (mainly salmonid fish). In practice, inclusion of population persistence relies largely on expert knowledge, spatial surrogates and general rules of thumb. For example, population dynamics can be incorporated using surrogates of species densities and core population ranges where abundances are known (Winston and Angermeier, 1995); where abundances are not known modelling habitat suitability and predicting probability of species occurrences may be used (Filipe et al., 2004; Brewer et al., 2007). Size and habitat complexity can also be used as a broad surrogate to increase the likelihood that a selected area supports the full range of habitat requirements and local connectivity to different habitats. These surrogates, however, are likely to be more effective at capturing critical habitat that is scattered throughout the river (e.g. riffles) than for capturing localized critical habitat (e.g. refuge pools maintained by groundwater in the

dry season). Size can also be used as a surrogate for the ability of an area to support large populations resistant to the effects of genetic and demographic stochasticity (Poiani et al., 2000). However, caution should be exercised in preferentially selecting large populations when source-sink population dynamics are suspected. For example, as little as 10 % of a population may be located in source habitats but be responsible for maintaining 90 % of the population found in sink habitats (Pulliam, 1988). In these instances, it may be best to maximize connectivity between populations (Freeman et al., 2007).

In designing a conservation area network, some of these surrogates of population persistence may be expressed as explicit targets – e.g. only consider populations of more than 50 individuals; conserve at least two populations of each species, preferably on different river systems. However, the lack of empirical data usually means that their incorporation is generally accomplished using qualitative statements of preference in the design phase, such as “larger populations are better”, or “where there is a choice, maximize connectivity” (Pressey et al., 2007).

Principle 4: Incorporate additional fixed spatial components of processes

Incorporating ecological integrity and connectivity cater for maintaining generic processes that are key drivers of the majority of freshwater ecosystems. There may also be instances where other specific natural processes are key determinants of the structure and functioning of freshwater ecosystems, and whose spatial components can be mapped. In conservation planning, these processes are referred to as "fixed spatial components" (Rouget et al., 2006) or "spatial catalysts" (Pressey et al., 2007). Some of these fixed spatial components may already have been identified in the three previous principles, e.g. areas that are strongly dependent on groundwater for maintaining spatial refugia may have been mapped in considering vertical connectivity. This principle addresses any additional fixed spatial components that may need incorporation.

In conservation planning, fixed spatial components are commonly defined using environmental surrogates such as climate, topography, geology, soils and vegetation. For example, high water yield areas could be identified based on a direct assessment of mean annual runoff. Where these data are not available, information on climate (e.g.

mean annual rainfall; evapotranspiration) and geology (to obtain an index of permeability) could be used (Rivers-Moore et al., in review). Identification of high water yield areas enables conservation planners to highlight specific areas where it is especially crucial to manage activities that are likely to have cascading impacts on the natural flow regime. This in turn prevents the disruption of a multitude of ecological, physical and chemical processes associated with altered flow regimes (Freeman et al., 2007).

USING THIS INFORMATION IN THE PLANNING PROCESS

The exercise of assembling all the information required to address these principles of persistence produces numerous GIS layers and rules, highlighting many areas that require management to support a wide range of processes. Two broad rules are proposed for incorporating this information into the design of a resource efficient, pragmatic freshwater conservation area network.

First, different levels of protection can be allocated to the areas flagged for conservation attention, depending on the specific function of the area and its sensitivity to human impacts. While conservation areas selected for representation are best managed in a high ecological integrity category, other areas (e.g. those for fish migration) may be able to withstand certain impacts. This notion is consistent with the hierarchy of freshwater protected areas proposed by Abell et al. (2007), in which “Freshwater Focal Areas” are embedded within “Critical Management Zones”, which in turn are embedded in “Catchment Management Zones”. Management of Freshwater Focal Areas is focussed largely on representation and is likely to be fairly restrictive, with diminishing restrictions in the latter two zones where the focus is largely on persistence. Using such a multiple-use zoning strategy helps to emphasize that not all conservation areas need to be “locked away” from human use. This in turn facilitates implementation because it strengthens the linkages between people and conservation, and therefore aligns more closely with integrated water resources management. Implementation can be further facilitated by matching these zones to existing planning categories used by land use planners and/or water resource managers in the planning region.

Second, persistence considerations should not be accomplished only at the end of a spatial design for representation, for example, by adding in all connected areas. Rather, the spatial design process should aim to achieve representation and persistence considerations simultaneously wherever possible, to maximise complementarity and efficiency of spatial design (Rouget et al., 2006; Sarkar et al., 2006). The concept of complementarity is well-established in systematic conservation planning (Pressey et al., 1993; Sarkar et al., 2006). Essentially, it seeks to maximise spatial efficiency by choosing areas that complement each other in meeting conservation goals. For example, where there is a choice of several areas for achieving representation of a specific biodiversity feature, an area should be chosen that contributes to as many other under-represented biodiversity features as possible. Alternatively, an area that is also going to be necessary for maintaining connectivity or spatially fixed components of processes should be chosen. Chapter 6 shows how an existing conservation planning algorithm can be applied in a freshwater setting to achieve complementarity in planning for representation and persistence simultaneously, while taking cognisance of a multiple-use zoning scheme.

In summary, applying these persistence principles for freshwater biodiversity generate an array of rules and spatial data that will be used at different stages in designing a spatially efficient conservation area network (Table 5.1). Some information will be incorporated before spatial design actually begins when formulating representation targets (e.g. fish population sizes and replication), developing ecological integrity filters (e.g. achieving pattern targets only in intact ecosystems), and defining planning units (e.g. using sub-catchments). Other information will be accommodated during spatial design (e.g. where choices for representation exist, favour upstream or downstream sub-catchments, or those containing fixed spatial components of processes). Finally, once complementary areas for representation and persistence have been selected, the design for connectivity can be completed (e.g. adding in remaining critical sub-catchments for maintaining longitudinal connectivity, or allocating riparian buffers to selected river systems).

Table 5.1 Consideration of persistence is incorporated throughout the steps used to design a conservation area network. The steps shown here are modified for freshwater settings from Gaston et al. (2002). Note that these steps are underpinned by an interactive process to facilitate effective implementation, a process which is not shown here (but see Knight et al., 2006).

Stages in designing a conservation area network	Persistence principle to consider
Define planning units	Connectivity through the use of sub-catchments
Compile data on biodiversity, current impacts and future threats, existing conservation initiatives	Ecological integrity
Record the extent of intact or restorable biodiversity per planning unit	Ecological integrity
Set quantitative conservation targets	Population persistence in setting population sizes, core ranges and rules for replication
Spatial design for both representation and persistence, aligning with existing conservation initiatives	Longitudinal connectivity; Vertical connectivity; Fixed spatial components of processes
Incorporation of any remaining critical management zones	Longitudinal connectivity; Lateral connectivity within sub-catchment (e.g. riparian zones and associated wetlands)

CONCLUSIONS

The framework presented here is broad enough to provide a useful starting point for collating the type of information needed to incorporate persistence into freshwater conservation planning. Planning for persistence is severely limited by our understanding of natural processes and requires the development of surrogate layers. Ultimately, these surrogates of persistence are hypotheses, which should be tested over time. Empirical data are needed to test whether these surrogates really promote persistence of biodiversity in the long term. For example, what do time series analyses reveal regarding the persistence of species and species assemblages within scenarios of degrading ecological integrity?

Planning for retention in the face of ongoing human threats to freshwater biodiversity is a critical component of freshwater conservation planning which has not been addressed here. It is not without its fair share of challenges. One of the most profound challenges in planning for retention will be mitigating anthropogenic disruption to the natural flow regime of the conservation area network. This will require the implementation of environmental flow recommendations and associated water allocations that maintain appropriate natural flow regime – variability, as well as quality and quantity of water. It is therefore critical that freshwater conservation plans and associated conservation actions are embedded into integrated water resource management decisions.

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CHAPTER 6.
PLANNING FOR REPRESENTATION AND
PERSISTENCE OF RIVER BIODIVERSITY:
A CASE STUDY FROM THE
OLIFANTS/DOORN WATER
MANAGEMENT AREA, SOUTH AFRICA

University of Cape Town

PLANNING FOR REPRESENTATION AND PERSISTENCE OF RIVER BIODIVERSITY: A CASE STUDY FROM THE OLIFANTS/DOORN WATER MANAGEMENT AREA, SOUTH AFRICA

ABSTRACT

The previous chapter presented four key principles to consider when planning for the persistence of freshwater biodiversity: selecting ecosystems of high ecological integrity; incorporating connectivity; incorporating populations that have a high probability of persisting; and identifying additional fixed spatial components of processes. The practicalities of gathering the data and conducting the conservation plan to address these principles are explored here using a case study in the Olifants/Doorn Water Management Area, South Africa. Spatial layers are developed for depicting ecological integrity, sub-catchment boundaries, riparian zones and wetlands, high water yield areas, and patterns of groundwater discharge and recharge. Although the planning region is relatively data-rich by global standards, several data-deficiencies were identified. Several environmental surrogates are suggested to address data-deficiencies, which should be combined wherever possible with existing field data and expert knowledge to improve the confidence in these surrogates. Methods are suggested for achieving spatial efficiency by simultaneously designing for representation and persistence of freshwater biodiversity. This study shows that complementarity can be addressed in different ways when using a conservation planning algorithm, depending on whether or not a multiple-use zoning strategy is used. For designing conservation area networks applicable to the real-world, achieving complementarity should therefore be evaluated within the context of multiple-use zones.

KEY WORDS: systematic conservation planning, freshwater, connectivity, integrated water resources management

INTRODUCTION

Early efforts in systematic conservation planning focussed largely on representing biodiversity pattern, such as species and ecosystem types, in a space efficient set of protected areas (Kirkpatrick, 1983). This strong focus on representation in protected areas – where biodiversity is represented, bounded and protected – is not particularly useful in the freshwater realm. Given the inherent connectivity of freshwater ecosystems, planning for the representation of biodiversity in isolated areas, without regard for upstream, downstream or upland areas, is conceptually flawed.

Even in terrestrial settings, the shortcomings of this static approach have been increasingly recognized and replaced with approaches that deal explicitly with both natural and threatening anthropogenic processes (see Pressey et al., 2007 for review). Approaches that consider natural processes can be grouped under the term “planning for persistence”. They originated from the growing realization that many natural processes responsible for maintaining and generating biodiversity pattern will not persist if they are not explicitly incorporated into spatial design. This is especially true for natural processes that operate over large areas, or require special spatial configurations, such as seasonal migration across large areas (Balmford et al., 1998; Cowling et al., 1999; Gaston et al., 2002; Rouget et al., 2006; Pressey et al., 2007). Conservation areas for representation are thus embedded into a conservation area network designed to maintain persistence of biodiversity. The term “planning for retention” can be used to group approaches that address threatening anthropogenic processes (Chapter 5; Cowling et al., 1999). Examples of such approaches include assessing the vulnerability of an area to likely future threats, scheduling conservation action to minimize loss of biodiversity in the face of ongoing threat, assigning appropriate conservation mechanisms and implementing conservation action (Gaston et al., 2002).

The progression of approaches in systematic conservation planning from representation to representation-and-persistence has greatly enhanced its potential applicability to planning for conservation in the freshwater realm. Chapter 5 combined emerging wisdom on planning for persistence in the terrestrial realm with concepts from freshwater ecology to derive four principles for incorporating persistence into

freshwater conservation planning: (1) selecting ecosystems of high ecological integrity; (2) incorporating connectivity; (3) selecting populations most likely to persist; and (4) mapping fixed spatial components of processes. The first two principles are persistence surrogates that cater for a range of natural processes that are key drivers of the structure and functioning of most freshwater ecosystems. The remaining two principles target more specific natural processes. If data are available for a particular species or species group, then these can be used to inform an assessment of which populations are likely to persist. Such data could include, for example, habitat requirements, minimum viable population sizes, or sensitivity to flow regime alteration. The final principle incorporates any additional persistence surrogates that have not already been captured, and which are fixed in space and can therefore be mapped. Examples include a map identifying areas of high water yield (Rivers-Moore et al., in review) that can serve to highlight areas that are particularly important for maintaining the natural flow regime, or mapping environmental gradients to provide maximum habitat complexity for a range of species (Rouget et al., 2006).

The above persistence framework was used in this study for developing a conservation area network for the Olifants/Doorn Water Management Area, South Africa. The study was commissioned by the national Department of Water Affairs and Forestry with the aim of informing water resources planning in the region, and hence had a strong focus on real-world application. In developing the conservation area network, this study aims to test the persistence framework in terms of its data requirements, applicability and value in designing a multiple-use conservation area network. It begins by collating the data required to address the persistence principles outlined in Chapter 5. These data are then applied, along with data on biodiversity pattern, to design a spatially efficient freshwater conservation area network. Finally, the merits of using a multiple-use zoning strategy during the design phase, as well as aligning conservation design with recommendations for future water resource development plans are discussed within the context of integrated water resources management.

METHODS

Study area

Biophysical characteristics

The Olifants/Doorn is one of 19 water management areas in South Africa (Figure 6.1), within which integrated water resources management and catchment management plans will be developed. It is a large area (approximately 56 750 km²), situated on the west coast of the country, and incorporates the entire drainage area of the Olifants River system, of which the Doring River is a major tributary. Smaller coastal river systems north and south of the Olifants River estuary are also included in the planning region.

Coastal lowlands rise to rugged mountains at almost 2000 m above sea level and climatic conditions differ considerably as a result of this variation in topography. Mean minimum temperatures in winter vary from -3 to 3 °C, whilst summer mean maximum temperatures range from 39 to 44 °C. This is a winter rainfall area, with mean annual rainfall varying between 100 to 1 500 mm across the planning region. Gross mean annual evaporation is high (approximately 1 500-2 200 mm). These steep environmental gradients give rise to one of the most diverse water management areas in South Africa with respect to its natural characteristics and water resources (DWAF, 2005a).

Although fish species richness is relatively low, the region is a notable southern African endemic hotspot for freshwater fish (Skelton et al., 1995). Nine out of the 12 indigenous freshwater fish species are endemic to the planning region, and all are threatened (Table 6.1). The Olifants River estuary is one of only three permanently-open estuaries on the west coast of South Africa and therefore represents a critical habitat for estuarine-associated fauna. The planning region contains one of the largest natural wetlands on the west coast of southern Africa (Verlorevlei), which is internationally recognised through its listing under the Ramsar Convention on Wetlands (<http://www.ramsar.org/>).

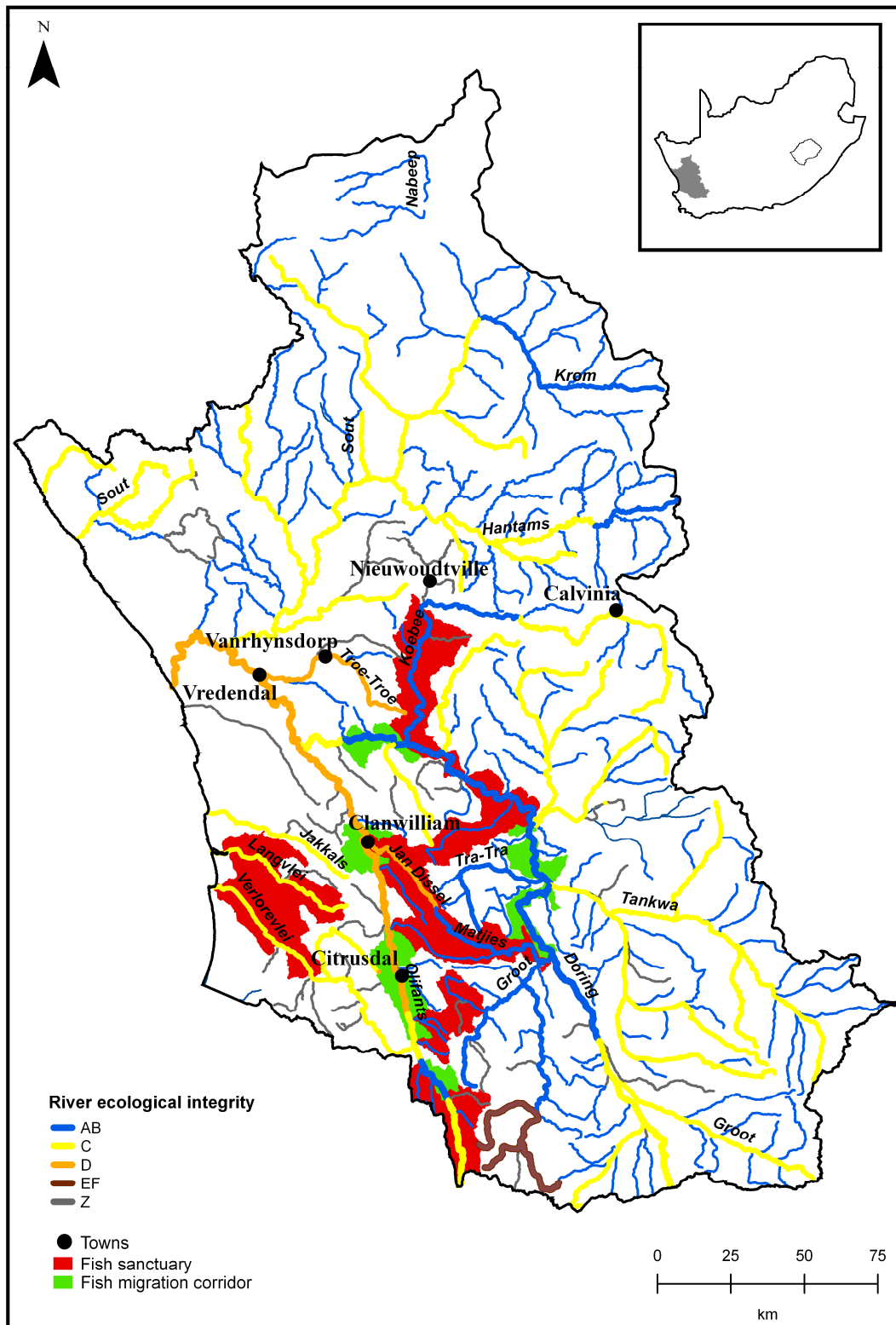


Figure 6.1 Study area showing major towns and location in South Africa, as well as ecological integrity, fish sanctuaries and fish migration corridors associated with the 1:500 000 river network. Main rivers and tributaries are indicated respectively by thick and thin lines

Groundwater plays a particularly important role in the region, sustaining river flow and refuge pools in the summer low flow periods (DWAF, 2005a). In addition, groundwater recharge in the region is believed to sustain coastal aquifers and groundwater dependent ecosystems some 100 km away.

Table 6.1 Freshwater fish species of the Olifants/Doorn. Common names marked with asterisks indicate species that are endemic to the planning region. Conservation status is based on a 2007 assessment (IUCN, 2007).

Common name	Scientific name	Conservation status
Verlorevlei redfin*	<i>Pseudobarbus burgi sp.</i>	Endangered
Fiery redfin*	<i>Pseudobarbus phlegethon</i>	Critically endangered
Clanwilliam redfin*	<i>Barbus calidus</i>	Vulnerable
Twee River redfin*	<i>Barbus erubescens</i>	Critically endangered
Clanwilliam sawfin*	<i>Barbus serra</i>	Endangered
Clanwilliam yellowfish*	<i>Labeobarbus capensis</i>	Vulnerable
Clanwilliam sandfish*	<i>Labeo seeberi</i>	Endangered
Spotted rock catfish*	<i>Austroglanis barnardi</i>	Endangered
Clanwilliam rock catfish*	<i>Austroglanis gilli</i>	Vulnerable
Chubbyhead barb	<i>Barbus anoplus</i>	Data deficient
Cape galaxias	<i>Galaxias zebratus</i>	Data deficient
Cape kurper	<i>Sandelia capensis</i>	Data deficient

Threats to freshwater biodiversity

Most of the land in the Olifants/Doorn is used as grazing for livestock, predominantly for sheep and goats. While land cover over much of the area is therefore largely unconverted, sheet erosion along water courses is prevalent (Fairbanks et al., 2000).

Although the area of irrigated land is small, irrigated agriculture is the economic mainstay of the region, with 95 % of total water use allocated to irrigation (DWAF, 2005a). Intensive production of deciduous fruits, citrus and grapes occurs along the lower reaches of the Olifants River, whilst large quantities of groundwater are abstracted for irrigation of potatoes on the west coast. The water resources of the Olifants River are largely regulated, while little development has taken place along the Doring River except for localised areas in the upper reaches. Tourism is an important and growing sector of the economy, and coastal towns suffer from water shortages over the summer tourist season due to peak demand. The conservation of freshwater

biodiversity in the region is dependent on balancing economic development with ecological water requirements, managing the following key threats: over-abstraction of both surface water and groundwater for irrigation purposes; impacts associated with invasive alien plant and fish species; degradation of wetland and riparian zones through the effects of grazing, bull-dozing and planting of crops; pollution from agricultural pesticides and the impacts of global climate change.

Relevant water resource management and conservation initiatives

An environmental flow assessment has been conducted for the Olifants and Doring Rivers and Olifants estuary in order to make recommendations for future development of water resources (Figure 6.2; Brown et al., 2006). The preferred environmental flow assessment scenario recommended increasing the water storage capacity of the region by raising the wall of an existing dam on the Olifants River (between sites 2 and 3 on Figure 6.2), as well as keeping water releases from the dam to a minimum in order to maximise water supply. Brown et al. (2006) suggest that although this compromises the ecological integrity of the Olifants River immediately downstream of the dam, it will have less of an overall environmental impact than expanding development to new areas of high ecological integrity, such as the Doring River. The recommendation also assumes maintaining the high ecological integrity of the Doring River, as well as major tributaries of the Olifants and Doring rivers. This will ensure adequate flow to the estuary (Figure 6.2), and support flow variability in the Olifants and Doring rivers.

Notable protected areas in the region include the Cederberg Wilderness Area, Groot Winterhoek Wilderness Area and Tankwa Karoo National Park. Mountain Catchment Areas, which recognize the importance of water supply areas, have also been declared for protection under the Mountain Catchment Areas Act (Act No. 63 of, 1970). In addition, two conservation planning initiatives, focussing on terrestrial ecosystems and their requirements, overlap with portions of the Olifants/Doorn: the Cape Action Plan for People and the Environment (Cowling et al., 2003) and the Succulent Karoo Ecosystem Programme (see <http://www.skep.org/>).

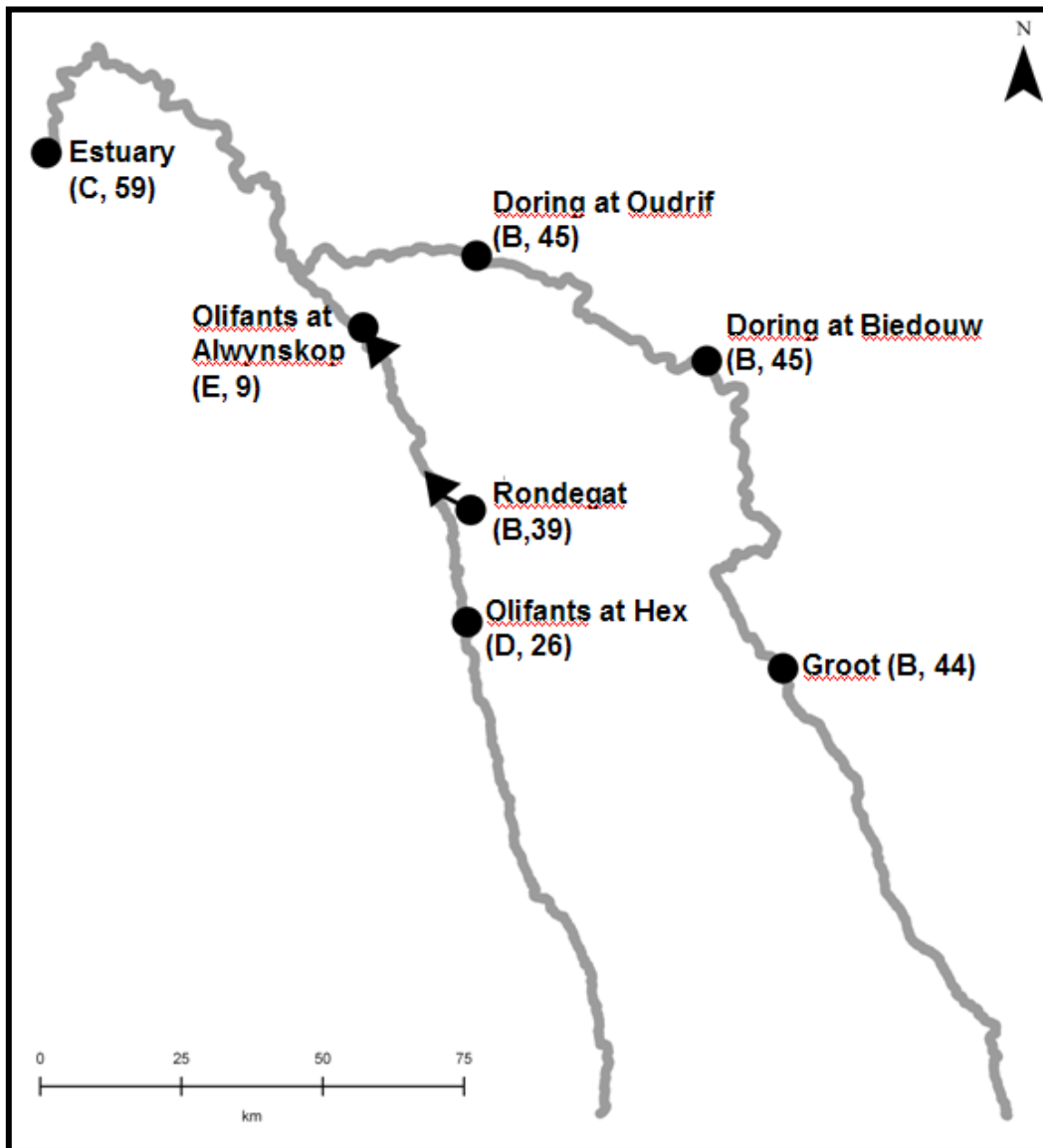


Figure 6.2 Sites for which environmental flow assessments were undertaken, after Brown et al. (2006). Sites are depicted by circles and dams by triangles. At each site, the number in brackets shows the percentage natural mean annual runoff required at the site to maintain the ecological integrity categories indicated by the letter in brackets. Ecological integrity categories are described according to Kleynhans (2000); see text for details. This also serves as a useful quantitative guide to the management of the required ecological integrity of many conservation areas selected in the Olifants/Doorn.

These planning initiatives catalysed the founding of the Greater Cederberg Biodiversity Corridor initiative, which explicitly addresses planning for persistence, embedding existing protected areas in the region within a multiple-use landscape that links interior and coastal habitats (<http://www.cederbergcorridor.org.za/>). Land stewardship outside protected areas forms a strong component of this initiative, and conservation priorities have been set to guide implementation of this component using a range of biodiversity and management criteria. The *ad hoc* inclusion of freshwater biodiversity concerns within this and other conservation initiatives in the region is more a consequence of the lack of systematic conservation planning for freshwaters than purposeful exclusion. Therefore, in addition to guiding water resources planning in the region, the opportunity also exists to expand the focus of existing conservation initiatives to include explicit consideration of freshwater biodiversity.

Expert knowledge

Extensive expert knowledge of the fauna, flora, river health and key threats to biodiversity exists in the Olifants/Doorn. This greatly facilitated collation of appropriate data for designing the freshwater conservation area network. This expert knowledge was harnessed in two ways. First, through individual consultations to collate available data and build consensus on how this information was to be used in the conservation plan. Second, through a workshop where experts were brought together to debate, review and refine the spatial data layers, collated in the Geographical Information System (GIS), ArcGIS 9 (ESRI, 2002).

Biodiversity pattern

Both species- and ecosystem-level surrogates were used for representing river biodiversity pattern, the elements of biodiversity that are static and can be mapped (Pressey et al., 2007). All 12 indigenous freshwater fish were used as the species-level surrogate of biodiversity (Table 6.1). Fish point locality records from the South African Institute for Aquatic Biodiversity (SAIAB), Albany Museum and CapeNature were

collated to produce a database of over 3000 localities. This was used by experts to inform the designation of fish sanctuaries for each species (see below).

Ecosystem-level surrogates were derived by combining three levels of information for all 1:500 000 rivers (DWAF, 2006) in the planning region: Level 2 ecoregions (Kleynhans et al., 2005), hydrological indices (Hannart and Hughes, 2003) and geomorphological river zones (Rowntree and Wadeson, 1999). Ecoregions are hierarchical units that broadly characterise the landscape through which a river flows, such that rivers in the same ecoregion share similar broad ecological characteristics to those in different ecoregions (Omernick, 1987). The hydrological index (HI) quantifies in a single statistic the amount and variability of water flow in a river and was used to distinguish three statistical river flow regimes (Dollar et al., 2006): permanent ($HI \leq 16.110$), seasonal ($16.110 < HI \leq 37.819$) and ephemeral ($37.819 < HI \leq 110$). The geomorphological river zones characterise the ability of each river reach to store or transport sediment, with each zone representing a different physical template available for biotic habitation. Moolman et al. (2006) stratified the slope profile of the river channel according to the descriptions and slope categories proposed by Rowntree and Wadeson (1999). These slope categories were grouped into four geomorphological river zones for this study: mountain streams, upper foothills, lower foothills and lowland rivers.

It was not possible to classify wetland types with a high level of confidence, and therefore wetlands were excluded from consideration of biodiversity pattern. However, wetlands were recognized for their functional role in the persistence of river biodiversity (see below).

Ecological integrity

River integrity was mapped for all 1:500 000 rivers using methods described in Chapter 4. This technique combines expert-derived ecological integrity categories for main rivers (Kleynhans, 2000) with modelled categories for tributaries. The latter uses the percentage of natural land cover from the 30 m resolution South African National Land Cover 2000 GIS layer (Fairbanks et al., 2000) as a surrogate for river integrity.

Ecological integrity for main rivers was mapped using three existing datasets: (1) present ecological status (Chapter 3; Kleynhans, 2000) based on an expert-derived assessment of six criteria (flow, inundation, water quality, stream bed condition, introduced instream biota, riparian or stream bank condition); (2) River Health Programme monitoring sites (River Health Programme, 2006) that use aquatic community and habitat indicators at a site level; and (3) aerial habitat integrity surveys at 5 km stretches along four rivers selected for environmental flow assessment (Brown et al., 2006). Present ecological status was used as the primary GIS layer, which has integrity categories ranging from A-F, where A is largely natural and F is unacceptably degraded (Kleynhans, 2000). For this study, rivers were considered intact if in an A or B integrity category, moderately modified if in a C category, and largely modified if in D-F categories. This was updated where necessary according to the latter two datasets. In instances where the condition of the river at the level of the landscape was better than that at the site level, experts were asked to review whether the differences were a result of localised impacts, or differences that occur at the landscape scale. The ecological integrity category was only updated if the difference was considered significant at a landscape scale: only one such refinement needed to be made (Jan Dissels River was downgraded from a C to a D ecological integrity).

Tributaries were considered intact if the minimum value for the percentage of natural land cover within the sub-catchment, 500 m and 100 m buffer of a river segment was $\geq 80\%$ and percentage erosion within a 500 m buffer of a river segment was $\leq 3\%$; remaining tributaries were regarded as not intact. Here, a river segment was defined as the portion of river between the 1:500 000 river confluences.

Connectivity

Longitudinal connectivity

Three aspects of longitudinal connectivity were incorporated: requirements for large migratory species, identification of free-flowing rivers, and selection of upstream management zones required to support river reaches selected for achieving representation. The requirements for large migratory species are discussed in the section

below on population persistence. For the second aspect, a free-flowing river was defined as an intact river, more than 100 km in length, that flows undisturbed from its source to its mouth, either at the coast, or at the confluence with a larger river, without encountering any dams, weirs or barrages and without being hemmed in by dykes or levees. This is similar to the WWF (2006) definition, but less stringent than that of Nilsson et al. (2005), which requires that mean annual flow has not been altered by more than 2 %. The final aspect for incorporating longitudinal connectivity was included during the design of the conservation area network. However, recognizing that it will be politically impossible (and not entirely necessary) to motivate for the inclusion of all rivers upstream of a conservation area, an *a priori* rule to select only those upstream areas that are the most critical for maintaining appropriate flows. In this way, all intact rivers having their source in areas of high water yield (see below) were considered critical to maintaining the present ecological integrity of downstream reaches.

Lateral connectivity

Lateral connectivity was broadly incorporated into conservation planning through modelling sub-catchments around each river segment in GIS (Arc Hydro, Version 1.1, ESRI, Redlands, CA) using 90 m resolution digital elevation data (US Shuttle Radar Topography Mission data; see <http://srtm.usgs.gov/>).

Important functional zones within sub-catchments were also identified by delineating riparian zones and wetlands. Three existing GIS layers were combined to map wetlands in the planning region: (1) the sensitive wetlands of the Western Cape Province (Shaw and de Villiers, 2001); (2) 1:50 000 perennial and non-perennial pans (Department of Land Affairs: Chief Directorate of Surveys and Mapping, 2005); and (3) delineations from the beta version of the national wetlands map (South African National Biodiversity Institute; see <http://wetlands.sanbi.org/>), derived from 30 m satellite imagery applied in conjunction with topography and wetness potential models to enhance wetland detection (Thompson et al., 2002; Ewart-Smith et al., 2006).

Riparian zones have not been comprehensively mapped for all 1:500 000 rivers in the planning region. Owing to limited resources, the study was unable to make use of

aerial photography, satellite imagery or field surveys for this aspect of lateral connectivity (but see Goetz, 2006; Goetz et al., in press). Instead, a buffer was applied to either side of all 1:500 000 rivers. The width of this buffer varied according to the geomorphological river zone. A buffer of 100 m was applied on either side of lower foothill and lowland river zones; whilst a 50 m buffer was used for the remaining zones. Buffer widths were based on expert experience regarding valley confinement and threat mitigation: lower foothills and lowland rivers are less confined and require wider buffers to mitigate the effects of agricultural practices (e.g. spraying of pesticides). These river buffers were applied around all rivers selected for representation or upstream management during the design of the conservation area network. It should be noted that application of river buffers in this context is intended to emphasize the importance of particular riparian areas in the conservation area network, and should not undermine the legal riparian buffer (32 m) that applies to all streams under the National Environmental Management Act (Act 107 of 1998).

Vertical connectivity

A predictive modelling approach using environmental surrogates was applied to map the probability of groundwater-surface water interaction (Conrad and Münch, 2006). This approach made use of six GIS layers believed to be the primary determinants of groundwater-surface water interaction within the region (Table 6.2). Each of these layers was classified into values between 0-4, which described their likelihood of groundwater-surface water interaction: absent, low, moderate, high and very high respectively. A weighting was also applied to each GIS layer, depending on its significance to groundwater interaction and the confidence in the data (Conrad and Münch, 2006; Table 6.2). Class values were multiplied by the associated weights assigned to each of the GIS layers, and then all GIS layers were summed to derive a composite map representing the probability of groundwater-surface water interaction. Probability scores were divided into three classes: high (scores > 9), medium (scores 5-9) and low (scores 0-4). Areas of high to medium likelihood of groundwater-surface water interaction were considered to be significant areas of groundwater discharge.

A map of groundwater recharge (mm per year) was derived from a nationally available GIS layer at a 1 km resolution (DWAF, 2005b). The method of determining groundwater recharge was based on the Chloride Mass Balance (Lerner et al., 1990), which applied a GIS model that replicates natural processes of direct groundwater recharge across the country, calibrated using known recharge values at several sites (DWAF, 2005b). Areas where groundwater recharge exceeds 30 mm per year were considered as having significant groundwater recharge in the South African context (DWAF, 2005b).

Fish population persistence

Three issues of population persistence were considered in designating fish sanctuary areas for the 12 indigenous freshwater fish species of the region. First, conservation targets stipulated that each species must be represented at least twice by populations that are preferably on different major river systems. Second, a relatively sound expert knowledge of the freshwater fish of the region allowed us to identify river reaches with the most suitable habitat and containing the largest populations for each species. Selection of these areas, hereafter fish sanctuaries, was guided by consideration of point locality records extracted from the aforementioned fish databases. Third, migration corridors were identified for those species requiring free passage between tributary and mainstem habitat. Fish sanctuary areas and migration corridors were combined for all species to provide a summary map of the areas required for representation and persistence of indigenous freshwater fish species of the region.

Additional spatial components of processes

Owing to the steep rainfall gradient, relatively small catchment areas contribute significantly to the water supply of the entire region. Areas of high water yield have already been delineated in the planning region as mountain catchment areas under South Africa's Mountain Catchment Area Act (Act No. 63 of, 1970). These were used to highlight areas that play a critical role in maintaining the natural flow regime of the region.

Table 6.2 GIS data layers used to map probability of groundwater-surface water interaction. After Conrad and Münch (2006).

GIS layer	Description	Rationale for use	Weight
Groundwater response units	Units that have similar hydrogeological characteristics. Based on 1:1 000 000 geology.	Units that depict boundaries between aquifer and non-aquifer geological formations. A significant change in permeability at these interfaces may result in groundwater discharging to the surface. A high weighting was assigned to this GIS layer, since geology plays a key role in groundwater characteristics.	3
Groundwater levels	Interpolated surface of depth to groundwater (m), based on borehole data.	Groundwater-surface water interaction is likely to be highest in areas where groundwater levels are shallow (i.e. close to the surface). A low weighting was assigned to this GIS layer because of the high uncertainty in the data.	1
Springs	The position of known springs in the planning region (not potential springs).	Points of known groundwater discharge. Springs in this area are important, therefore this GIS layer received a high weighting.	3
Geological faults	The position of geological faults in the landscape. Based on 1:250 000 geological structures.	Faults are often favourable flow paths for groundwater, although there are many faults that are weathered and essentially sealed, with no associated groundwater presence or movement. For this study, it was assumed that all faults are water bearing and a high weighting was assigned.	3
Groundwater dependent ecosystems	Probability of occurrence of groundwater dependent ecosystems. Based on 1:250 000 vegetation groupings.	Management of groundwater in the immediate vicinity of these ecosystems is crucial. A moderate weighting was applied to this GIS layer due to its coarse national scale.	2
Groundwater contribution to baseflow	Based on monthly flow data at the scale of a quaternary catchment.	This GIS layer is the most commonly used national indicator of groundwater-surface water interactions. For much of the planning region, however, these data indicate no groundwater-fed baseflow, yet field experience indicates groundwater is an important contributor to maintaining these systems during the dry season (pers. comm., C. Brown, 2006). The GIS layer was consequently assigned a low weighting.	1

Conservation area network design

Chapter 5 emphasized the need to plan for both representation and persistence simultaneously in order to design a spatially efficient conservation area network. The concept of complementarity – where a set of areas are chosen to complement, rather than duplicate, each other in the conservation objectives they achieve – forms the computational backbone of most systematic conservation planning algorithms (Sarkar et al., 2006).

The MARXAN conservation planning algorithm was used to aid decisions regarding the trade-off between multiple conservation objectives (Ball and Possingham, 2000; Possingham et al., 2000), and was adapted for use in a freshwater setting. MARXAN was used in conjunction with the CLUZ interface (Smith, 2005) to view the results in Arcview (Version 3.2, ESRI, Redlands, CA). MARXAN is a complementarity-based algorithm that uses a simulated annealing optimization method to achieve conservation targets at least cost (Ball & Possingham, 2000). This approach combines iterative improvement with occasional acceptance of changes that make the conservation area networks more costly. The algorithm begins by creating a conservation area network based on randomly selected planning units. It then iteratively tests other potential designs by randomly adding and removing planning units, seeking to reduce the “combined cost” of the conservation area network. The combined cost depends on the sum of three measures: the planning unit cost, the target penalty and the boundary cost (Box 1). These measures enable MARXAN to find the best possible solution (lowest cost solution) by exploring the trade-offs between multiple conservation objectives as planning units are iteratively added and removed. These costs (Box 1) therefore provide a means of measuring complementarity and efficiency in planning for representation and persistence.

Processing of data for MARXAN

Using sub-catchments as planning units, and input GIS data on river ecological integrity, river types and fish sanctuaries, the extent of intact river types within each planning unit was quantified and loaded into MARXAN, as well as the presence/absence of a fish sanctuary. Conservation targets were loaded into MARXAN for each river type as 20 % of the total length of each river type. For fish sanctuaries, the conservation target in MARXAN was 100 % (fish sanctuaries had already been identified according to a conservation target of at least two populations per fish species, preferably on different river systems). A very high target penalty factor (Box 1) was set for all features in order to encourage full representation of all river types and fish species.

Because connectivity tends to be non-directional in terrestrial settings, the MARXAN boundary cost (quantified as the length of the boundary; Box 1) is usually applied to all boundaries to favour connectivity. The boundary cost needs to be refined for planning in the freshwater realm to accommodate the directional connectivity of lotic systems. This was achieved by applying a boundary cost only to those boundaries belonging to pass-through sub-catchments, defined as those sub-catchment boundaries that intersected a 1:500 000 river. All boundaries were assigned a uniform boundary cost of 200 (irrespective of length). This value was derived using a series of MARXAN scenarios to test the relative importance of reducing the boundary penalty compared to the planning unit cost: setting the boundary penalty too low produced a relatively scattered solution, while setting the boundary penalty too high resulted in the selection of many connected sub-catchments that did not contribute toward conservation targets (e.g. sub-catchments in which river systems were not intact).

A planning unit cost was also applied to each sub-catchment in MARXAN so that where choices existed between sub-catchments with similar biodiversity features and spatial connectivity, preference would be given to sub-catchments (1) identified as important for spatially-fixed persistence surrogates; or (2) aligned to existing conservation initiatives. The former were defined as those where the extent of significant groundwater discharge and recharge areas, and significant water yield areas together was > 50 % of the sub-catchment. The latter were defined as those where

> 50 % of the sub-catchment was either under formal protection or had been identified by the Greater Cederberg Biodiversity Corridor initiative as a priority for consolidation of existing protected areas. Here, the South African protected areas GIS layer (Reyers *et al.*, 2007) and farm boundaries identified by the Greater Cederberg Biodiversity Corridor were used, respectively. A uniform value was assigned to all sub-catchments as the baseline planning unit cost and then all sub-catchments qualifying under criteria (1) and (2) were “discounted” to less than this baseline value. These values were determined through a series of MARXAN scenarios to test sensitivity to varying the planning unit cost and associated discount.

Design protocol

Nine steps were then used to design a conservation area network for representation and persistence of freshwater biodiversity:

1. Exclude river ecosystems below the Olifants River from the areas available for representation, based on the recommendations for future water resources development (Brown *et al.*, 2006).
2. “Earmark” all sub-catchments containing free-flowing river reaches prior to beginning MARXAN. “Earmarking” these sub-catchments forces their inclusion in the conservation area network.
3. Run MARXAN to select a spatially efficient configuration of sub-catchments that achieves the residual conservation targets from Step 2. Use the boundary penalty and planning unit costs determined from the sensitivity analyses.
4. Select remaining sub-catchments required to maintain downstream conservation areas selected in Steps 2 and 3. Restrict selections to only those upstream areas critical to sustaining environmental flow recommendations, in line with the rule to allow for some small-scale water resource development in the region.
5. Select remaining sub-catchments required to support migration between fish sanctuaries.
6. Select all wetlands associated with sub-catchments selected in Steps 2-5.
7. Select areas of high groundwater discharge and recharge, and high water yield areas.

8. Where conservation targets could not be achieved in intact systems, assess feasibility of restoring appropriate river reaches, guided by expert-derived data available for the country at a quaternary catchment scale (Best Attainable Ecological Management Class; Kleynhans, 2000), as well as the judgement of river practitioners in the planning region.
9. Zone according to Table 6.3: rivers and their associated buffers were allocated to Freshwater Focal Areas if their sub-catchments were selected in Steps 2 and 3, Critical Management Zones if selected in Steps 4-7, or Critical Restoration Zones if selected in Step 8. All remaining areas identified in Step 7 were assigned to a Critical Management Zone, and then all sub-catchments selected in Steps 2-8 were flagged as Catchment Management Zones

Table 6.3 Zones allocated to the spatial components comprising the Olifants/Doorn conservation area network. Zones are based on a hierarchical protection strategy for freshwaters in which Freshwater Focal Areas are embedded within Critical Management Zones, which in turn are embedded in Catchment Management Zones (Abell, Allan & Lehner, 2007). Management of Freshwater Focal Areas is focussed largely on representation and is likely to be fairly restrictive, with diminishing restrictions in the latter two zones where the focus is largely on persistence.

Zone	Spatial component*
Freshwater Focal Area	River reach and buffer selected for achieving conservation targets
Critical Management Zone	Free-flowing river reach Upstream river reach and buffer critical for supporting a downstream Freshwater Focal Area River reach and buffer required to support migration between fish sanctuaries Wetland supporting a Freshwater Focal Area Significant areas of groundwater-surface water discharge Significant areas of groundwater recharge Significant water yield area
Catchment Management Zone	Sub-catchment containing Freshwater Focal Areas and/or Critical Management Zones
Critical Restoration Zone	Sub-catchment where feasible restoration will result in improved achievement of river type conservation targets

* A river segment is defined as the portion of river between confluences of a 1:500 000 river; this is also the segment around which sub-catchments (planning units) are delineated

RESULTS

Biodiversity pattern and ecological integrity

The combination of 15 Level 2 ecoregions, three river flow regimes, and four geomorphological river zones produced 78 distinct river types (Figure 6.3). In total, 34 sub-catchments were selected as fish sanctuaries (Figure 6.1). Conservation targets for *Barbus erubescens* were lowered to one occurrence because it is a highly localised endemic that can only be represented in one system. Only 57 % of rivers in the planning region are considered intact, with main rivers proportionally more impacted than tributaries (Figure 6.1).

Connectivity

In terms of longitudinal connectivity, the Doring River was identified as one of the few remaining large free-flowing rivers in South Africa. All intact western and eastern tributaries of the Olifants and Doring rivers respectively have their source in high water yield areas (Figure 6.4) and were thus flagged as likely upstream management zones that are critical to sustaining any downstream conservation areas subsequently selected on the Olifants and Doring rivers, as well as the Olifants estuary.

A total of 528 sub-catchments, averaging approximately 110 km² in size, were modelled as planning units. Just over 2 500 wetlands were mapped for consideration of lateral zones of importance within sub-catchments, ranging in size from < 1 km² to approximately 117 km², and totalling almost 530 km². The buffered river network within the conservation area network highlighted just over 690 km² of riparian zone that would need to be managed to ensure the persistence the associated river biodiversity.

Significant areas of groundwater discharge (Figure 6.4a) cover approximately 7 600 km² (13 %) of the planning region, while significant areas of groundwater recharge in the South African context (DWAF, 2005b) comprise 3 350 km² (6 %; Figure 6.4a).

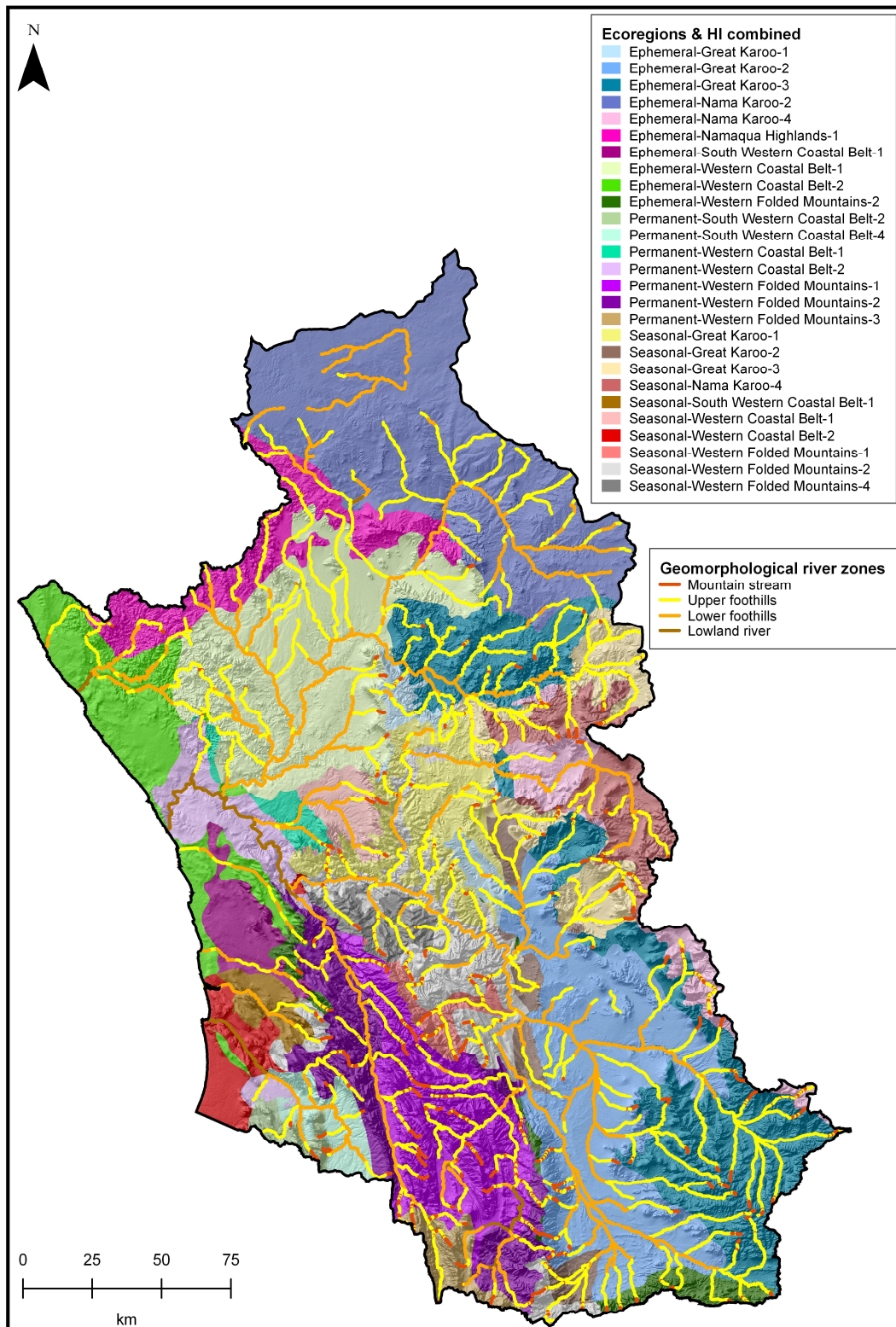


Figure 6.3 River types of the Olifants/Doorn. The shaded landscape polygons show unique combinations of Level 2 ecoregions and hydrological index classes. These were combined with geomorphological river zones to derive 78 distinct river types.

Fish population persistence

In addition to the criteria for replication that were considered during fish sanctuary designation, 15 sub-catchments were identified as fish migration corridors between tributary and mainstem habitat of fish sanctuaries (Figure 6.1). Importantly, linkages were excluded for many of the smaller-sized species, where artificial barriers have protected tributary populations from predation by invasive alien fish species prevalent in the mainstem rivers of the region.

Additional spatial components of processes

Areas of significant water yield are associated with the high rainfall mountainous areas of the Cederberg and overlap to some extent (32 %) with the significant areas of groundwater recharge (Figure 6.4b).

Conservation area network design

The final conservation area network requires that 342 (65 %) of the sub-catchments in the Olifants/Doorn be afforded some level of conservation management (Figure 6.5): 24 % as Freshwater Focal Areas where human activities should be restricted to ensure the maintenance of current natural condition, 41 % as Critical Management Zones required to maintain natural processes that support the integrity of the Freshwater Focal Areas; and 3 % as Critical Restoration Zones.

Within sub-catchments, 45 % of the buffered 1:500 000 river network needs to be managed as a Freshwater Focal Area, 12 % as Critical Management Zones, and 5 % as Critical Restoration Zones (Figure 6.5). In addition, a large proportion of the planning region (19 %) was flagged as a Critical Management Zone for management of wetlands and groundwater that support the persistence of Freshwater Focal Areas. Conserving all Freshwater Focal Areas will result in the achievement of conservation targets for all fish species and 72 % of the river types. With feasible restoration of the Critical Restoration Zones (Figure 6.5), 86 % of the river types will be able to meet their conservation targets. It is not possible to fully achieve the conservation targets of the remaining river types, as restoring natural examples of these river types is not feasible in the planning region given the levels of degradation and human use.

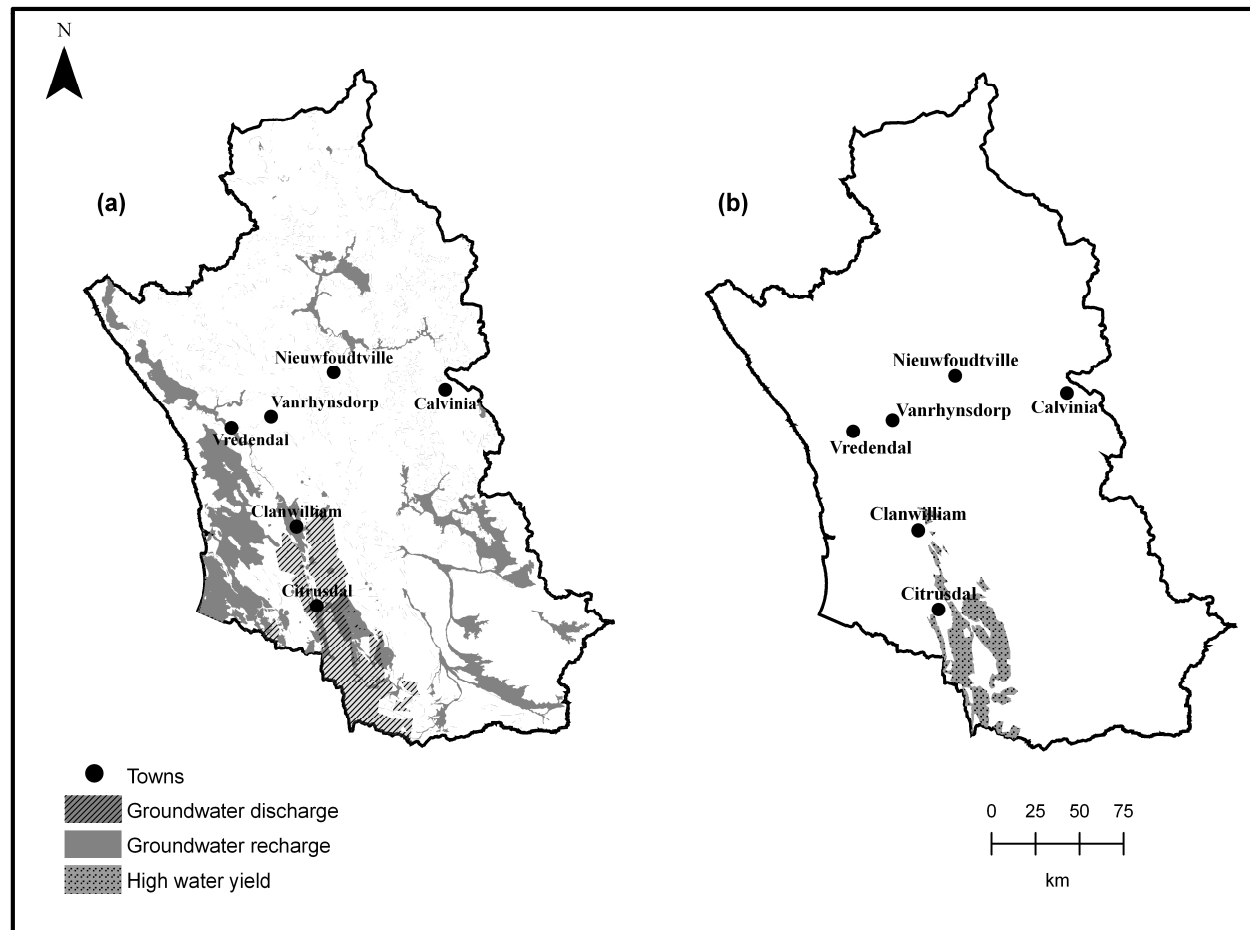


Figure 6.4 Significant areas of (a) groundwater discharge and recharge; and (b) water yield in the planning region.

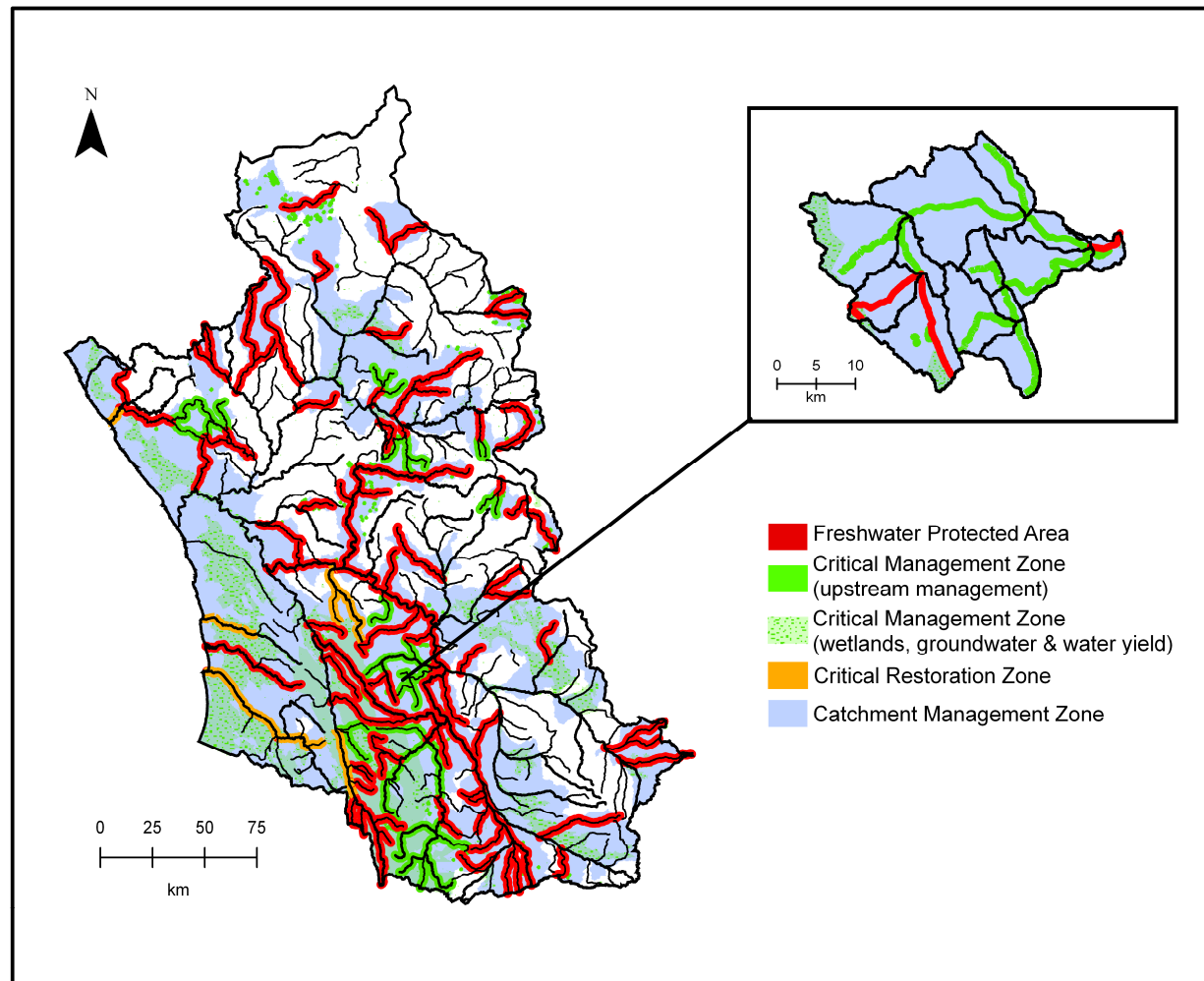


Figure 6.5 Conservation area network designed to support representation and persistence of freshwater biodiversity in the Olifants/Doorn. Management zones are comprised of spatial components listed in Table 6.3. The insert provides an example of the zones allocated within sub-catchments based on the different spatial components.

DISCUSSION

The Olifants/Doorn, with its low human settlement and relatively intact ecosystems compared to the rest of the country, offers relatively good opportunities for conserving freshwater ecosystems. The Doring River is the second largest free-flowing river in the country (Chapter 4). Since options are extremely limited for conserving large, free-flowing rivers in South Africa, it should therefore be regarded as a strategic national priority for conservation. In addition, this region is a southern African endemic hotspot for freshwater fish, and the establishment, restoration and management of recommended fish sanctuaries should receive priority attention. Finally, the cumulative impact of small-scale abstraction of both surface water and groundwater (particularly during the critical spawning season) needs to be addressed if biodiversity within the conservation areas is to persist in the long term. By mapping groundwater discharge and recharge patterns, and areas of high water yield, the conservation area network highlights the areas most critical to begin managing in this regard.

This study used rapid desktop methods to assemble the data required by the persistence framework in Chapter 5, and applied these in combination with data on biodiversity pattern to design a freshwater conservation area network. It set out to test the persistence framework in terms of (1) data requirements, (2) applicability and (3) value in designing a multiple-use conservation area network. Each of these is discussed below under separate headings.

Data requirements

The persistence principles outlined in Chapter 5 provide a good starting point for incorporating freshwater biodiversity persistence into conservation plans. However, no single recipe exists for collating the data since every planning region will differ in the level of information and expert knowledge available, as well as in the characteristics that drive natural processes in the region. For example, in the Olifants/Doorn, freshwater fish in seasonal rivers are strongly dependent on groundwater. Mapping areas of significant groundwater discharge and recharge highlights areas that are key

drivers of persistence in the region. In planning regions that are less dependent on groundwater, this may not be an issue.

Even in the Olifants/Doorn, which is relatively data-rich by global standards, several data-deficiencies were identified (e.g. mapping ecological integrity of tributaries, delineating riparian zones and identifying significant areas of groundwater discharge). Environmental surrogates were used to address these data-deficiencies, and confidence in these surrogates was improved by combining them wherever possible with existing field data and expert knowledge. Empirical studies are needed to test the validity of these assumptions and improve guidelines for managing the long term persistence of freshwater biodiversity. Such studies may include, for example, ground-truthing of wetlands and significant areas of groundwater discharge, testing assumptions about thresholds in land cover disturbance and alterations to the natural flow regime, or testing the contribution of intact riparian zones to persistence.

Future work to improve the confidence in desktop classification of wetland types would also broaden the conservation plan to include representative samples of freshwater systems other than rivers. This is necessary, as preliminary analyses indicate that river types do not serve as good surrogates for wetland types and vice versa (Nel et al., 2006).

Achieving complementarity in representation and persistence

Planning for both representation and persistence simultaneously allows an evaluation of the complementarity of an area in terms of a range of conservation objectives. While this is a widely accepted concept in terrestrial conservation planning (Sarkar et al., 2006), it has only recently been applied in freshwater conservation planning (Linke et al., 2007; Moilanen et al., 2008). Conservation planning algorithms, such as MARXAN, offer several options for achieving complementarity between representation and persistence, such as achieving connectivity through applying boundary penalties (Box 1), or beginning the MARXAN runs by “earmarking” areas required for spatially fixed components of processes and then achieving residual representation targets.

Some persistence issues, such as longitudinal connectivity, are dependent on the location of conservation areas for biodiversity pattern representation. In this study,

applying MARXAN's boundary penalty to pass-through sub-catchments aided the selection of connected sub-catchments while achieving conservation targets for biodiversity pattern. Although this only achieved partial longitudinal connectivity, it permitted the allocation of multiple-use zones allowing a distinction to be made between sub-catchments required for achieving conservation targets (Freshwater Focal Areas) and those that were important for maintaining longitudinal connectivity, but that were not essential for achieving conservation targets (Critical Management Zones). In instances where a multiple-use zoning strategy is not used, an alternative method that allows simultaneous consideration of representation and connectivity is to use complementarity-based algorithms that are linked to rules that automate the selection of upstream and downstream linkages (Linke et al., 2007; Moilanen et al., 2008).

There are two ways of achieving complementarity between representation and spatially fixed components of processes (i.e. not dependent on the location of conservation areas for representation). One way is to force selection of these areas in the subsequent design by earmarking them prior to running the conservation planning algorithm (Rouget et al., 2006). This is the method followed for sub-catchments associated with free-flowing rivers: seven river type targets were immediately met through this action. Constraining selection in such a way inevitably results in a less spatially efficient design, but the benefits gained in managing persistence issues may make this a sensible compromise. However, it may not be worth compromising spatial efficiency if the spatially fixed persistence surrogates are to be zoned to incorporate less restrictive uses than those for representation. In this case, another method would be to favour these areas during planning for representation using the planning unit cost. Here, planning unit cost is traded off against the cost of spatial efficiency in representation, so that discounted planning units will be favoured where there are choices that result in similar spatial efficiency. This was the method applied in the case of the significant areas of groundwater discharge and recharge, and high water yield areas.

In summary, when using a conservation planning algorithm, the manner in which complementarity between persistence and representation is achieved depends on whether or not a multiple-use zoning strategy will be applied during design. Systematic conservation plans generally defer zoning to the implementation phase, although there

are conservation planning algorithms, such as MarZone (MARXAN with Zones – developed by Ian Ball, Matthew Watts and Hugh Possingham), that explicitly acknowledge multiple-use zones during spatial design. Currently, however, MarZone operates on the basis of locking areas into *a priori* zones. Adapting this algorithm to evaluate the options for assigning areas to different zones based on achieving complementarity for representation and persistence holds potential.

Using a multiple-use zoning strategy

The decision on whether to use a multiple-use zoning strategy during the design phase should be guided by the relative challenges and advantages this presents. The key challenge is that, as with the persistence principles, a conceptual multiple-use zoning strategy only provides a starting point for assigning levels of protection. These zones need to be refined to incorporate aspects such as site- and region-specific threats and overall sensitivity of the associated biodiversity to these threats. Simply, broad multiple-use zones are surrogates for the level of protection required for an area and need to be tested during the course of implementation. While there is wide recognition of the problems with the use of surrogates (see Rodrigues and Brooks, 2007 for review), they are accepted as a necessary tool for aiding systematic conservation planning (Pressey, 2004). Thus, the argument of ignoring the use of zones until the implementation phase seems to be somewhat unjustified.

Incorporating persistence criteria generally creates a space-hungry plan. To consider the advantage of using multiple-use zones imagine Figure 6.5 as showing all zones as Freshwater Focal Areas. The allocation of multiple-use zones in the Olifants/Doorn, where water resource development is seen as a socio-economic priority, helps to make this conservation area network politically more “palatable”, emphasizing that strong use restrictions do not necessarily apply to all areas. This, in turn, greatly facilitates the implementation process.

Implementation can be further supported by relating the zones to existing land use and water resource planning categories (Knight et al., 2006). This study was undertaken with the specific goal of informing water resources planning in the region. In making water resource decisions, water resource planners attempt to balance human

and ecosystem needs by examining scenarios of desired future condition of rivers (much like the ecological integrity categories in Table 6.2). Preparing a freshwater conservation plan, and preferably relating the zones to the desired future condition categories at a sub-catchment level, enables explicit consideration of freshwater biodiversity needs during this process.

In addition, the explicit and systematic inclusion of freshwater biodiversity into existing conservation initiatives in the region can be achieved by relating the multiple-use zones of Figure 6.5 to planning categories relevant to these initiatives. Conservation actions within these initiatives complement existing water resources planning in the area by focussing on land management within the conservation area network. Here, delineations of multiple-use zones within the sub-catchment, such as wetlands, riparian and groundwater zones (Figure 6.5) are particularly pertinent.

CONCLUSIONS

This study demonstrates that it is possible to collate and apply the data required to address the persistence principles outlined in Chapter 5, although these depend on using a range of surrogates, which should be rigorously examined over time. This study also shows that achieving complementarity between representation and persistence during the design phase can be addressed in different ways, depending on whether or not a multiple-use zoning strategy is used. This, in turn, is likely to influence the final conservation area network design. Making use of a multiple-use zoning strategy in freshwater conservation planning can facilitate implementation because (1) it strengthens the linkages between people and conservation; and (2) the concept aligns more closely with planning categories used by water resource managers and land use planners. Given these implementation practicalities, zones should be applied in the design phase, rather than merely at the end once the design is complete. To further support the effective implementation of this conservation plan, the entire exercise needs to be nested within a larger social process of stakeholder engagement that evolves over time, fostering a spirit of cooperation, alignment of mandates, co-learning and adaptive management. The formation of catchment management forums within the water management area under the South Africa's Water Act (Act 36 of 1998), along with the

imminent development of a catchment management strategy for the region, offers an ideal vehicle for mainstreaming a conservation vision into water resources management of the Olifants/Doorn Water Management Area.

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University of Cape Town

CHAPTER 7.
SYNTHESIS

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SYNTHESIS

This thesis has made a novel contribution to both systematic conservation planning and integrated water resources management by developing a suite of frameworks and practical applications that can be used for planning in freshwater settings. These frameworks were developed to address the need for a more systematic, strategic, coordinated and landscape-level approach to planning for freshwater biodiversity in both the water and conservation sector. It began by consolidating a wide range of disparate literature applicable to freshwater conservation planning, drawing on disciplines such as terrestrial conservation planning, integrated water resources management, freshwater ecology, hydrology, geomorphology and spatial technology. It then developed frameworks for:

- Rapid assessment of freshwater ecosystem endangerment and protection levels (Chapter 3);
- Expansion of protected area systems for freshwater biodiversity (Chapter 4);
- Planning for the persistence of freshwater biodiversity (Chapter 5); and
- Designing a conservation area network that incorporates both representation and persistence of freshwater biodiversity within a multiple-use landscape (Chapter 6).

By developing common conservation frameworks around which the water and conservation sector can engage and debate, this thesis seeks to enhance the integration of freshwater biodiversity concerns into both these sectors. The frameworks are generic and can be applied to other regions of the world. Key findings and recommendations varied in terms of their relevance within the South African and international context, and are discussed below under the five original aims outlined in the introduction.

KEY FINDINGS AND RECOMMENDATIONS

1. Identifying the basic requirements for undertaking freshwater conservation planning (Chapter 2)

Using principles drawn from a wide range of disciplines, six broad requirements for undertaking conservation planning in freshwater settings were distinguished (Chapter 2; Table 2.1). An important conclusion of this synthesis was that the requirements for freshwater conservation planning are similar to those encompassed in existing terrestrial conservation planning procedures. However, the specific methods for dealing with each of these differ, particularly in relation to connectivity. Freshwater-specific recommendations were therefore made for dealing with these requirements, based on a synthesis of recent methods that have been used.

2. Developing systematic methods for assessing river ecosystem endangerment and protection levels at broad sub-continental scales (Chapter 3)

The state of freshwater ecosystems in South Africa is dire, far worse than that of terrestrial ecosystems (Chapter 3; Figure 7.1). This is consistent with the worldwide trend in freshwater biodiversity, and supports the numerous calls for increased conservation efforts (Abell, 2002; Saunders et al., 2002; Dunn, 2003; Dudgeon et al., 2006). Quantifying the state of freshwater ecosystems, especially concurrent to assessments of terrestrial, marine and estuarine ecosystems, was extremely useful in highlighting the strategic need for elevating freshwater biodiversity concerns on the agendas of both national and local conservation organizations. In addition, the development of spatially explicit outputs (Figures 3.3 and 3.5) and comparative summary statistics (Figure 7.1) had a compelling visual impact on decision makers, much higher than previous presentation in tabular or report format (Reyers et al., 2007).

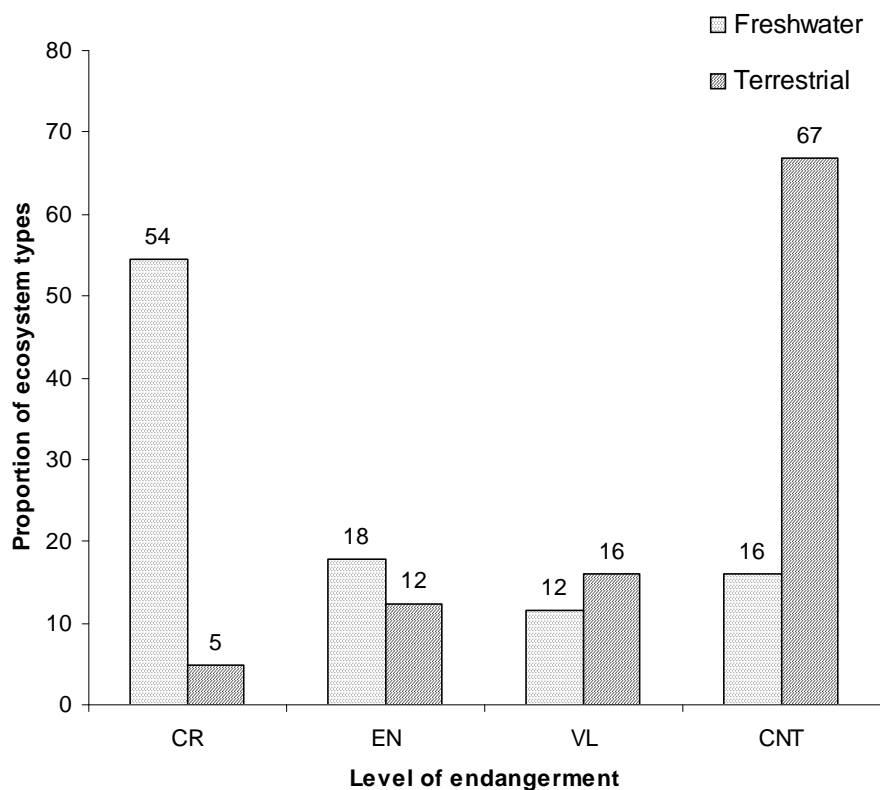


Figure 7.1 Comparative levels of endangerment for (a) terrestrial (Driver et al., 2005) and (b) river (Chapter 3) ecosystem types in South Africa, where CE = critically endangered; E = endangered; V = vulnerable and CNT = currently not threatened. Proportion of ecosystems is expressed as a percentage of the total number of ecosystems in each endangerment category. Total number of terrestrial and river ecosystem types is 438 and 112 respectively.

Similar to experiences in terrestrial conservation planning (Margules and Pressey, 2000), deciding on the appropriate spatial scale of analysis for freshwater conservation plans is important in terms of the questions that are being addressed. Chapter 2 addressed endangerment and protection levels at a national, or sub-continental, scale. In this instance, dealing with large rivers provided useful contextual data for a broad national audit of the overall state of biodiversity. However, the results in Chapter 2 had limited value in identifying high value conservation areas at a local level of planning. In tackling local levels of planning in Chapters 4 and 6 it was necessary to include a modelled integrity assessment of the tributaries of these large rivers, at a level at which individual river reaches are usually managed in South Africa.

Despite several years of river health monitoring in South Africa, quantitative data are generally available for large rivers only (e.g. River Health Programme, 2006). The reason for this is logical – river health monitoring focuses on measuring cumulative impacts over entire catchments. Monitoring sites are therefore generally established on mainstem rivers which cumulatively reflect the impacts in the tributaries. However, conservation planning at the local level requires an assessment of the integrity of individual tributaries as well as cumulative impacts. Expanding river health monitoring sites to the tributaries would help rectify this disparity, thus better aligning conservation and monitoring efforts.

The distinction that had to be made between main rivers and tributaries – owing to a lack of integrity data – nevertheless highlighted the importance of tributaries for conservation. Main rivers are usually heavily utilized and regulated to improve water security for social and economic use (Chapter 3), while tributaries are often less impacted than main rivers (Figure 4.3). Tributaries could be viewed as refugia for river biodiversity, replenishing other parts of the river system from time to time (Freeman et al., 2007). For this replenishment to occur, main rivers need to be managed in a state that supports connectivity; however, this need not necessarily translate to a largely natural state (Thieme et al., 2007). For example, migratory fish may be able to tolerate migration corridors in a moderately modified mainstem river. This realization was important for allocating multiple-use zones in Chapter 6.

3. Devising systematic methods for addressing the freshwater biodiversity gaps prevalent in protected area systems (Chapter 4)

It is widely acknowledged that partial inclusion of rivers in protected areas is no guarantee for their protection since they are often impacted by threats outside the protected area (Saunders et al., 2002). This is supported by the finding that only 50 % of the rivers within protected areas can be considered intact (Chapter 3). However, this is a higher proportion compared to outside protected areas, where only 28 % of the rivers are intact. Consistent with calls from Abell et al. (2007), this finding suggests that protected area systems can play a positive role in the overall

conservation of freshwater ecosystems, and should not be dismissed as a conservation strategy. With more attention to design and management (Chapters 4 and 6), protected areas have the potential to become powerful cornerstones for catalyzing conservation action at a catchment level. Several criteria and recommendations are made in Chapter 4 on locating and designing protected areas that would benefit river biodiversity. Establishing large wilderness areas, strategically located around free-flowing rivers, is a particularly important priority to pursue in expanding protected area systems (Chapter 4). This is also a widely supported strategy for terrestrial biodiversity (Terborgh and Soulé, 1999).

Deriving a river network topology is essential to the automation of connectivity analyses (Figure 4.1). This is a coding that describes a tree-hierarchy of river segments, reflecting the flow of water. It enables automatic identification of river segments upstream and downstream of any particular locality, groups river segments of the same river reach, and also informs the delineation of networked sub-catchments in GIS. Chapter 4 used this topology to enable the assessment of pattern and process related issues (e.g. representation within river segments, connectivity of river reaches within protected areas, and identification of free-flowing rivers). Chapter 6 used the topology to assign boundary costs to all pass-through sub-catchments to achieve complementarity between representation and connectivity.

4. Developing a framework for incorporating persistence into the design of freshwater conservation area networks (Chapter 5)

This persistence framework sets out four principles for planning for the persistence of freshwater biodiversity. Conventional approaches to water ecosystem management, such as river health assessment and monitoring, are founded on the concept of ecological integrity (Karr et al., 1986), which can be viewed as a key knowledge-bridge between freshwater ecology and systematic conservation planning. Selecting ecosystems of high ecological integrity serves as a filter for ensuring the capture of functional processes when selecting areas for conservation

(Chapter 5). Consequently, it is the first principle suggested in a framework for planning for the persistence of freshwater biodiversity. Addressing issues of connectivity in spatial design is the second principle (Chapter 5; Pringle, 2001). The remaining two principles target more specific natural processes through incorporating population persistence issues where information on particular species and populations are available, and through mapping fixed spatial components of processes through the use of environmental surrogates, such as mapping high water yield areas that are critical for maintaining the natural flow regime (Chapter 5).

The ideal way to incorporate connectivity in a plan for freshwater biodiversity is to select entire catchments upstream of the area, with downstream reaches also selected in the case of migratory animals. This may work for small catchments, headwater sub-catchments or catchments that are extremely remote from human populations. However, it is generally not feasible to place entire catchments under a uniformly high level of protection. In practice, it becomes necessary to enter the exercise of balancing human and ecological concerns. Here, plans need to identify conservation areas that contain the biodiversity that needs to be represented, and embed these in a network that incorporates the areas that are the most critical to manage to ensure that the biodiversity within these conservation areas persist. Making use of a multiple-use zoning strategy (e.g. Abell et al., 2007) becomes a practical option in these instances, highlighting that different levels of protection (and hence utilization) can be afforded to areas within the conservation area network.

5. Testing the application of this persistence framework to planning for both representation and persistence of freshwater biodiversity at a local scale relevant to integrated water resources management (Chapter 6)

Planning for the representation and persistence of freshwater biodiversity should be considered simultaneously to ensure complementarity and efficiency in design (Chapter 6). While this is a widely accepted concept in the context of terrestrial conservation planning (Sarkar et al., 2006), it has only recently made its way into

freshwater conservation planning (Chapter 6; Linke et al., 2007; Moilanen et al., 2008). Complementarity between representation and connectivity is particularly important in freshwater settings. The conservation planning algorithm, MARXAN, can be refined for consideration of directional connectivity in freshwater settings by applying boundary costs only to pass-through sub-catchments (Chapter 6). This method seeks to achieve optimal connectivity while selecting areas for representation. It produces a partially connected design for representation – the design for connectivity then needs to be manually completed through the incorporation of additional critical areas required for connectivity but not for representation (Chapter 6). The advantage of this approach, over approaches that apply strict connectivity rules to select connected sub-catchments, is that it is possible to distinguish between representation and connectivity areas, and on this basis assign appropriate zones. Complementarity between representation and spatially fixed components of processes (e.g. free-flowing rivers, groundwater discharge and recharge areas) can be incorporated in two ways – either by forcing representation in these areas (*sensu* Rouget et al., 2006), or by applying cost surfaces that favour these areas where there are choices for representation (Chapter 6). The former method is more suitable when the spatial component of persistence is to be zoned with the same use restrictions as areas selected for representation; the latter is a better option if use restrictions are less severe.

It is acknowledged that the study area in which these concepts were tested is relatively data rich area by global standards, and that the specific methods may need to be refined in data poor areas. However, there are certain generic findings regarding the need for multiple-use zoning. These findings show that complementarity can be achieved in different ways when using a conservation planning algorithm, depending on whether or not a multiple-use zoning strategy is used (Chapter 6). This, in turn, is likely to influence the final conservation area network design. Given the practicalities of multiple-use zoning at local levels of planning, zones should be applied in the design phase, rather than merely at the end once the design is complete.

FUTURE RESEARCH

- The challenges of implementing freshwater conservation plans are immense (Chapter 2). Research to support this process includes: (i) Exploring appropriate policy mechanisms that address the policy gap between water ecosystem management and conservation. This seems to be a worldwide phenomenon that needs attention – organizations are responsible either for biodiversity or for water provisioning, but seldom both. Early lessons emerging from efforts to improve cross-sector policy integration and cooperation in South Africa (Roux et al., in press) suggest that this will require both informal and formal mechanisms. (ii) Developing cooperative conservation frameworks that address the level of cooperation required to achieve optimal conservation outcomes: When is a high level of cooperation appropriate? What does it cost? What conditions are necessary for it to exist? What benefits can it realistically generate? Lessons can be drawn from related research in the health sector (Ansari and Phillips, 2004), but remain to be tested within the context of freshwater conservation; (iii) Developing social learning processes whereby stakeholders, policy-makers, scientists and practitioners can learn together and improve conservation practices; and (iv) The systematic collation of working principles from the emerging evidence base in freshwater biodiversity conservation.

- Directed empirical research in freshwater ecology has much to offer for improving the scientific rigour of freshwater conservation plans. These include empirical studies exploring: thresholds to land cover change; thresholds to flow alteration; setting of non-uniform population and ecosystem conservation targets; likely responses of river systems and associated biota to global climate change; and the relationships of species and species assemblages within surrogate ecosystem classifications.

- Applying a multiple-use zoning strategy is a far more challenging exercise than simply selecting whole catchments. It requires making tough decisions, often with very little empirical foundation: Which upstream areas are the most critical to

sustaining downstream flows and water quality? Which streams can we compromise for human use? What are the minimum flow requirements to sustaining the conservation areas and the biodiversity they support (in terms of variability, quantity and quality)? Where should land use activities be most restricted? Many of these questions have been the subject of debate in the field of environmental flow assessment for almost a decade (Tharme, 2003). Although most of the research in this field has focused on large, heavily utilized and perennial rivers, a generic desktop approach that uses hydrological classification methods combined with ecological calibration holds potential for providing rapid environmental flow guidelines around the world in the coming decade (Arthington et al., 2006). The integration of these approaches to inform the design and subsequent management of conservation area networks holds immense potential.

- Frameworks presented in this thesis need to be extended to include representation of wetland biodiversity. Research focussed on planning for wetland biodiversity is slowly emerging (Ausseil et al., 2007; Hamilton et al., 2007), however, this remains to be integrated with planning for river biodiversity. Such integration introduces a whole new dimension of connectivity refinements for conventional conservation planning algorithms. Seeking an efficient spatial design will require assessing the trade-offs between complementarity of river and wetland biodiversity pattern, together with longitudinal connectivity of selected areas, together with lateral connectivity of selected areas. This is a complex, but necessary requirement for moving from river conservation plans to freshwater conservation plans.

- There have been several recent calls for better integrative planning across terrestrial, freshwater and marine realms (Sloane et al., 2007; Vance-Borland et al., 2008). The freshwater-specific frameworks developed here set the scene for meaningful interaction with terrestrial and marine conservation planners regarding the best way forward for more integrative planning. Developing both technical and implementation frameworks for integration across terrestrial, freshwater and marine

realms is an exciting next step towards a more holistic approach to conserving the Earth's biodiversity.

CONCLUSIONS

This thesis has shown that:

- There is a strong need to close the gap that exists between the water resource and conservation sectors – this includes gaps in research, policy-making and practice.
- Conservation planning frameworks developed in the terrestrial realm can be applied to freshwater settings. Developing freshwater-specific methods within these frameworks (as was done in this thesis) will help to some extent in closing this gap.
- A necessary step in closing the gap between the water resource and conservation sectors is to incorporate biodiversity visions into integrated catchment management strategies. Conservation planning products will therefore need to be tailored so that they can inform water resources planning and management.
- Multiple-use zoning cannot be ignored in freshwater conservation planning given the need to tailor conservation planning for water resources planning and management, as well as the utility of water to humans and the strong connectivity of freshwater systems (which often produces a space hungry conservation plan).
- Co-development of a biodiversity vision with catchment managers, using systematic conservation planning frameworks such as those in this thesis, is also crucial to closing the gap between the water resource and conservation sectors..

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APPENDIX 1

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Appendix 1. Main river ecosystems in South Africa, with their associated ecosystem status and protected area category. The river ecosystem name is described in terms of the geomorphic province through which it flows, and a number which indicates the hydrological index class into which it falls. The hydrological index describes flow variability of the river, where regions of low flow variability (commonly containing perennial-type rivers) have a hydrological index class close to 1, and the semi-arid regions of high flow variability (commonly containing periodic- or ephemeral-type rivers) would be assigned to classes 6-8. Ecosystems labelled “N/A” under their protected area category have no remaining intact main river.

River ecosystem name	Ecosystem status	Protection level
Bushveld basin 1	Critically Endangered	Not protected
Bushveld basin 2	Critically Endangered	Hardly protected
Bushveld basin 3	Endangered	Hardly protected
Bushveld basin 4	Critically Endangered	N/A
Bushveld basin 5	Critically Endangered	N/A
Bushveld basin 6	Currently Not Threatened	Poorly protected
Bushveld basin 7	Critically Endangered	Hardly protected
Cape fold mountains 1	Vulnerable	Hardly protected
Cape fold mountains 2	Endangered	Poorly protected
Cape fold mountains 3	Endangered	Poorly protected
Cape fold mountains 4	Critically Endangered	Hardly protected
Cape fold mountains 5	Critically Endangered	Poorly protected
Cape fold mountains 6	Critically Endangered	Not protected
Cape fold mountains 7	Critically Endangered	N/A
Great escarpment 1	Vulnerable	Moderately protected
Great escarpment 2	Endangered	Poorly protected
Great escarpment 3	Endangered	Not protected
Great escarpment 4	Critically Endangered	Not protected
Great escarpment 5	Critically Endangered	N/A
Great escarpment 6	Endangered	Not protected
Great escarpment 7	Critically Endangered	N/A
Highveld 1	Critically Endangered	Not protected
Highveld 2	Critically Endangered	Hardly protected
Highveld 3	Critically Endangered	Not protected
Highveld 4	Critically Endangered	Not protected
Highveld 5	Critically Endangered	Not protected
Highveld 6	Critically Endangered	N/A
Highveld 7	Critically Endangered	Hardly protected
Kalahari basin 3	Critically Endangered	N/A
Kalahari basin 7	Currently Not Threatened	Hardly protected
Kalahari basin 8	Currently Not Threatened	Well protected
Karoo 1	Vulnerable	Not protected
Karoo 2	Critically Endangered	N/A
Karoo 3	Critically Endangered	Poorly protected
Karoo 4	Endangered	Hardly protected
Karoo 5	Endangered	Not protected
Karoo 6	Currently Not Threatened	Not protected
Karoo 7	Currently Not Threatened	Not protected
Lebombo 1	Endangered	Moderately protected
Lebombo 2	Vulnerable	Moderately protected
Lebombo 3	Currently Not Threatened	Well protected
Lebombo 4	Currently Not Threatened	Not protected
Lebombo 7	Currently Not Threatened	Well protected
Lebombo 8	Currently Not Threatened	Well protected

River ecosystem name	Ecosystem status	Protection level
Lesotho highlands 1	Vulnerable	Well protected
Lesotho highlands 2	Critically Endangered	Hardly protected
Lesotho highlands 3	Endangered	Not protected
Lesotho highlands 4	Critically Endangered	N/A
Limpopo flats 2	Critically Endangered	Not protected
Limpopo flats 3	Critically Endangered	Not protected
Limpopo flats 4	Critically Endangered	Poorly protected
Limpopo flats 6	Vulnerable	Not protected
Limpopo flats 7	Vulnerable	Hardly protected
Lower Vaal and Orange valleys 2	Critically Endangered	N/A
Lower Vaal and Orange valleys 3	Endangered	Not protected
Lower Vaal and Orange valleys 4	Critically Endangered	N/A
Lower Vaal and Orange valleys 5	Critically Endangered	N/A
Lower Vaal and Orange valleys 6	Vulnerable	Not protected
Lower Vaal and Orange valleys 7	Endangered	Not protected
Lower Vaal and Orange valleys 8	Critically Endangered	N/A
Lowveld 1	Critically Endangered	Poorly protected
Lowveld 2	Critically Endangered	Poorly protected
Lowveld 3	Critically Endangered	Not protected
Lowveld 7	Vulnerable	Well protected
Lowveld 8	Currently Not Threatened	Well protected
Mpumalanga highlands 1	Endangered	Poorly protected
Mpumalanga highlands 2	Critically Endangered	Moderately protected
Namaqualand highlands 3	Vulnerable	Not protected
Namaqualand highlands 5	Critically Endangered	Not protected
Namaqualand highlands 6	Endangered	Not protected
Namaqualand highlands 7	Currently Not Threatened	Not protected
Namib 3	Critically Endangered	N/A
Namib 4	Critically Endangered	N/A
Namib 5	Critically Endangered	Not protected
Namib 6	Critically Endangered	Not protected
Namib 7	Endangered	Not protected
Polokwane plain 1	Critically Endangered	N/A
Polokwane plain 2	Critically Endangered	N/A
Polokwane plain 3	Critically Endangered	N/A
Polokwane plain 5	Critically Endangered	N/A
Polokwane plain 6	Currently Not Threatened	Not protected
Polokwane plain 7	Endangered	Not protected
South-eastern coastal hinterland 1	Endangered	Poorly protected
South-eastern coastal hinterland 2	Vulnerable	Hardly protected
South-eastern coastal hinterland 3	Vulnerable	Poorly protected
South-eastern coastal hinterland 4	Critically Endangered	Poorly protected
South-eastern coastal hinterland 5	Critically Endangered	N/A
Southern coastal platform 1	Vulnerable	Poorly protected
Southern coastal platform 2	Critically Endangered	Not protected
Southern coastal platform 3	Critically Endangered	Not protected
Southern coastal platform 4	Critically Endangered	N/A
Southern Karoo 4	Critically Endangered	N/A
Southern Karoo 5	Critically Endangered	Not protected
Southern Karoo 6	Critically Endangered	N/A
Southern Karoo 7	Critically Endangered	N/A
Soutpansberg 1	Critically Endangered	Not protected
Soutpansberg 2	Endangered	Well protected
Soutpansberg 3	Critically Endangered	N/A

River ecosystem name	Ecosystem status	Protection level
Soutpansberg 7	Currently Not Threatened	Moderately protected
Swartland 1	Critically Endangered	N/A
Swartland 2	Critically Endangered	Poorly protected
Swartland 3	Critically Endangered	Poorly protected
Swartland 4	Critically Endangered	N/A
Swartland 5	Critically Endangered	N/A
Waterberg 1	Critically Endangered	N/A
Waterberg 2	Endangered	Poorly protected
Waterberg 3	Endangered	Poorly protected
Waterberg 4	Currently Not Threatened	Well protected
Zululand coastal plain 1	Currently Not Threatened	Well protected
Zululand coastal plain 2	Currently Not Threatened	Not protected
Zululand coastal plain 3	Currently Not Threatened	Well protected
Zululand coastal plain 4	Currently Not Threatened	Poorly protected

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