River conservation planning: accounting for condition, vulnerability and connected systems

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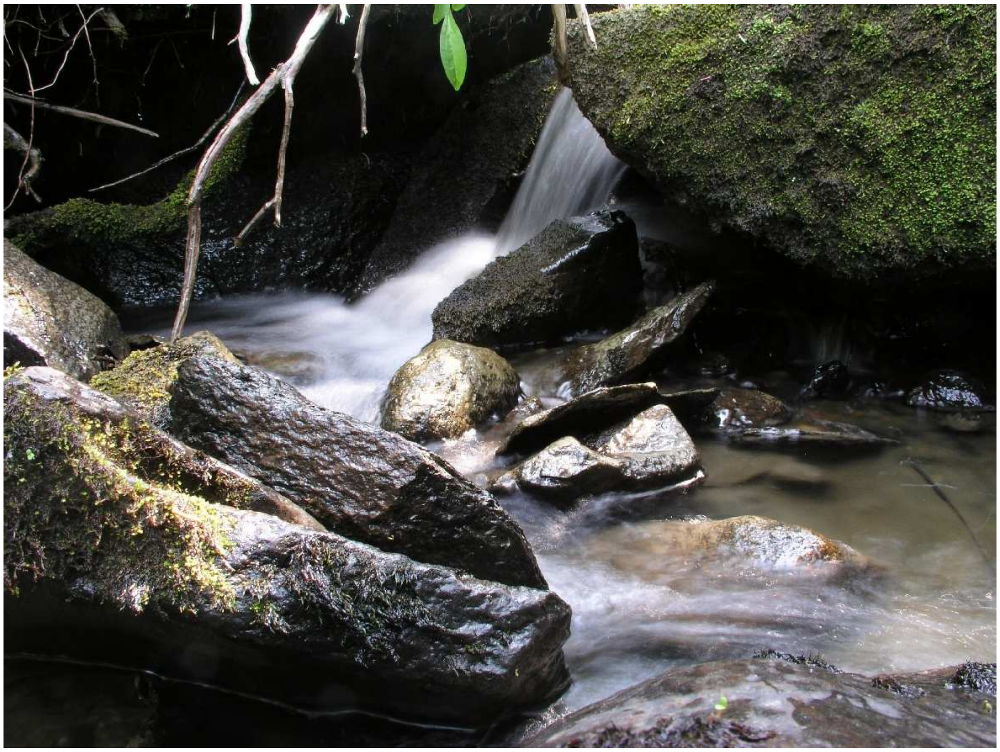

this thesis is dedicated to my late grandparents

josef oettl

for teaching me curiosity

elsa oettl

for teaching me compassion

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Abstract

Conservation science in rivers is still lagging behind its terrestrial and marine counterparts, despite increasing threats to freshwater biodiversity and extinction rates being estimated as five times higher than in terrestrial ecosystems. Internationally, most protected rivers have been assigned reserve status in the framework of terrestrial conservation plans, neglecting catchment effects of disturbance. While freshwater conservation tools are mainly index based (e.g. richness, rarity), modern terrestrial and marine conservation planning methods use complementarity-based algorithms - proven to be most efficient at protecting a large number of taxa for the least cost. The few complementarity-based lotic conservation efforts all use broad river classifications instead of biota as targets, a method heavily disputed in the literature. They also ignore current condition and future vulnerability.

It was the aim of this thesis to develop a framework for conservation planning that:

a) accounts for the connected nature of rivers

b) is complementarity based and uses biota as targets

c) integrates current status and future vulnerability

I developed two different approaches using macroinvertebrate datasets from Australia, Canada and the USA. The first new method was a site/based two-tiered approach integrating condition and conservation value, based on RIVPACS/AUSRIVAS – a modelling technique that predicts macroinvertebrate composition. The condition stage assesses biodiversity loss by estimating a site-specific expected assemblage and comparing it to the actual observed assemblage. Sites with significant biodiversity loss are flagged for restoration, or other management actions. All other sites progress to the conservation stage, in which an index of site-specific taxonomic rarity is calculated. This second index (O/E BIODIV) assesses the number of rare taxa (as defined by <50% probability of occurrence). Using this approach on a

dataset near Sydney, NSW, Australia, I was able to identify three regions: 1) an area in need of restoration; 2) a region of high conservation value and 3) an area that had high conservation potential if protection and restoration measures could counteract present disturbance.

However, a second trial run with three datasets from the USA and Canada highlighted problems with O/E (BIODIV). If common taxa are predicted at lower probabilities of occurrence ($p<50\%$) because of model error, they enter the index and change O/E (BIODIV). Therefore, despite an attractive theoretical grounding, the application of O/E (BIODIV) will be restricted to datasets where strong environmental gradients explain a large quantity of variation in the data and permit accurate predictions of rare taxa. It also requires extensive knowledge of regional species pools to ensure that introduced organisms are not counted in the index.

The second approach was a proper adaptation of terrestrial complementarity algorithms and an extension to the Irreplaceability-Vulnerability framework by Margules and Pressey (2000). For this large-scale method, distributions for 400 invertebrate taxa were modeled across 1854 subcatchments in Victoria, Australia using Generalised Additive Models (GAMs). The best heuristic algorithm to estimate conservation value was determined by calculating the minimum area needed to cover all 400 taxa. Solutions were restricted to include rules for the protection of whole catchments upstream of a subcatchment that contained the target taxon. A summed rarity algorithm proved to be most efficient, beating the second best solution by 100 000 hectares. To protect 90% of the taxa, only 2% of the study area need to be protected. This increases to 10% of the study area when full representation of the targets is required.

Irreplaceability was calculated by running the heuristic algorithm 1000 times with 90% of the catchments randomly removed. Two statistics were then estimated: f (the frequency of selection across 1000 runs) and *average c* (contribution to conservation targets). Four groups of catchments were identified: a) catchments that have high contributions and are always selected; b) catchments that have high contributions and are not always selected; c) catchments that are always chosen but do not contribute many taxa; d) catchments that are rarely chosen and did not contribute many taxa. *Summed c*, the sum of contributions over 1000 runs was chosen as an indicator of irreplaceability, integrating the frequency of selection and the number of taxa protected.

Irreplaceability (I) was then linked to condition (C) and vulnerability (V) to create the ICV-framework for river conservation planning. Condition was estimated using a stressor gradient approach (SGA), in which GIS layers of disturbance were summarised to three principal axes using principal components analysis (PCA). The main stressor gradient – agriculture – classified 75% of the study area as disturbed, a value consistent with existing assessments of river condition. Vulnerability was defined as the likelihood that land use in a catchment would intensify in the future. Hereby current tenure was compared to land capability. If a catchment would support a land use that would have a stronger effect on the rivers than its current tenure, it was classified as vulnerable. 79% of catchments contained more than 50% vulnerable land.

When integrating the three estimators in the ICV-framework, seven percent of catchments were identified as highly irreplaceable but in degraded condition. These were flagged for urgent restoration. Unprotected, but highly irreplaceable and highly vulnerable catchments that were still in good condition made up 2.5% of the total area. These catchments are prime candidates for river reserves.

The ICV framework developed here is the first method for systematic conservation planning in rivers that is complementarity-based, biota-driven but flexible to other conservation targets and accounts for catchment effects, thus fulfilling all the gaps outlined in the aims.

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Chapter 1. Introduction: The case for new approaches in freshwater conservation

1.1. Background

In recent years, the state of freshwater ecosystems and the loss of biodiversity within them has gained more attention from scientists, managers and the public than ever before. Leading scientists have called for increased efforts, not only regarding the traditional disciplines of river health assessment and ecological integrity, but also, specifically, conservation of freshwater biodiversity (Moss, 1999; Cullen, 2003; Dunn, 2003; Fitzsimons & Robertson, 2005). At the same time, both government initiatives and non-government organisations have declared a vested interest in new methods for the conservation of freshwater systems (Abell, 2002; Saunders *et al.*, 2002; Kingsford *et al.*, 2005).

This surge in interest in freshwater conservation follows a golden age of river health assessment in the 1980s and 1990s. On almost every continent, new biological and physical assessment methods were developed, replacing the dated Saprobien system and related biotic indices (Kölkwitz & Marsson, 1909). Following the development of methods like RIVPACS (River Invertebrate Prediction and Assessment System, Wright *et al.*, 1993a; Clarke *et al.*, 2003), IBI (Index of Biotic Integrity, Karr, 1999), AUSRIVAS (AUStralian RIVer Assessment System, Simpson & Norris, 2000) and BEAST (BEnthic Assessment of SedimenT, Reynoldson et al., 1997) on respective continents, large-scale assessments were conducted. While concern about loss of taxon richness and biological diversity had been raised previously (Lake, 1980; Dudgeon, 1992; Hughes & Noss, 1992; Allan & Flecker, 1993), these assessments on a country or even continental scale give a quantitative measure of the degradation.

These large-scale assessments showed similar outcomes in different corners of the globe. While a comprehensive national assessment of wadeable streams is being conducted in the USA (USEPA, 2004), Bryce *et al.* (1999) found that 60 percent of streams in the Mid-Atlantic highlands were at high risk of degradation – in some ecoregions as much as 90 percent. While

England and Sweden have performed national assessments of river health (Wright *et al.*, 1993a; Wiederholm & Johnson, 1997; Wright *et al.*, 1998), Australia has carried out the only continental programs of river assessment. After data had been collected for the National River Health Program (Schofield & Davies, 1996; Davies, 2000), a national audit - the Assessment of River Condition (ARC, Norris *et al.*, 2001; Norris *et al.*, in press) - was conducted on the rivers in Australia’s intensive land-use zone. This national audit classified only 14 percent of the assessed rivers as largely unmodified – 86 percent were as degraded. This figure was backed up by the Victorian Index of Stream Condition (Ladson et al., 1999) – a state-wide assessment that assessed 75-80 percent as degraded. `

Another result of recent intensive surveys is information about the status of particular taxa. A high percentage (72 percent) of freshwater mussels in the USA have been reported as endangered or threatened according to the IUCN redlist (Williams *et al.*, 1993a; Abramovitz, 1996; Abell *et al.*, 2000; World Conservation Union, 2000), which is especially alarming as 30 percent are endemic. Moyle (1995) reports 63 percent of freshwater fishes of California as threatened. Taxon specific assessments have not been conducted in Australia with similar precision, however some studies indicate similar rates of loss. Horwitz (1990) reports 57 freshwater crustacean species as being endangered and one as extinct. Cullen & Lake (1995) describe 12 fish species as endangered and one extinct. Worldwide, species extinction in freshwater environments is estimated higher than in terrestrial ecosystems (McAllister *et al.*, 1997; Abell, 2002). Ricciardi & Rasmussen (1999) estimate the extinction rate at four percent per decade, which is five times the terrestrial value.

One of the reasons why the extinction rate in rivers may be higher than in terrestrial systems is that conservation efforts might be more difficult: not only are local effects are felt in river ecosystems, but disturbances elsewhere in the catchment can contribute to loss of biodiversity. In the preface to his 2004 review article about landscape influences on rivers,

J. David Allan quotes Leopold *et al.* (1964): “Rivers are the gutters down which run the ruins of continents”. While this is a rather drastic way to put it – Noel Hynes (1975) preferred ‘The valley rules the stream in every respect’ - a look at recent Australian data illustrates the statements. While for 47 percent of Australian rivers the local habitat condition remained unmodified, two other measures of catchments effects influenced river condition (Norris *et al.*, 2001; Norris *et al.*, in press). The catchment disturbance index, which includes land use in the catchment as well as infrastructure (e.g., roads, powerlines, settlements), classified only 15 percent of the assessed rivers as largely unmodified. The nutrient and sediment load index – comparing pre-European and actual loads of nutrients and sediments - showed over 90 percent of streams in degraded condition. While many terrestrial conservation programs only consider immediate threats at a site, the above example highlights the need to develop a conservation program that acknowledges the connected nature of rivers.

1.2. Objectives and properties of a proposed system for biodiversity assessment and conservation planning

In a recent appeal for increased protection of aquatic ecosystems, Cullen (2003) suggests three measures to maintain biodiversity in rivers:

1. Identification and protection of important areas of habitat in a reserve system
2. Reduction of specified threatening processes on particular species across the landscape
3. Restocking of threatened species to try and re-establish natural communities.

My thesis will focus mainly on the first point – identification of systems of high conservation value and their protection in a whole-catchment context.

While some rivers are protected in the framework of terrestrial reserves (labelled insufficient by Lake, 1980; Maitland, 1985; Skelton *et al.*, 1995), few of these have been specifically designed as freshwater reserves. Notable exceptions are the Nahanni National Reserve in Canada (Saunders et al., 2002) and the Pacaya-Samiria National Reserve in Peru (Bayley et al., 1991). Apart from the obvious – terrestrial conservation planning does usually not consider aquatic taxa as targets, which are thus underrepresented (Nilsson & Gotmark, 1992) – specific threats to freshwater ecosystems like the catchment degradation mentioned above are not considered in the reserve selection (Angermeier & Winston, 1999; Filipe *et al.*, 2004). A number of authors has recently called for modifying the more advanced terrestrial and marine conservation planning techniques and modifying them for use in freshwater systems (Fitzsimons & Robertson, 2005; Kingsford *et al.*, 2005).

Starting in the 1970s, terrestrial scientists moved away from assigning reserves based on aesthetic values (Neel & Cummings, 2003) or availability (Pressey et al., 2000). Led by Ratcliffe (1971), managers and scientists were trying to conserve a maximum number of taxa, ecosystems or ecological processes. Discussed in detail in Chapter 2, most early assessments were either based on richness, rarity or diversity metrics. Kirkpatrick (1983, p. 128) showed that a selection process based on raw metrics was inefficient:

> “A major drawback of a listing of priority areas on the basis of a single application formula is that there is no guarantee that the priority area second or third on the list might not duplicate the species, communities or habitats that could successfully be preserved in the first priority area”.

This seminal study was the birth of modern assessments of conservation value, in which the principle of complementarity plays the central role. Complementarity-based selection algorithms in conservation theory look for areas that add as many under-represented surrogates (freshwater taxa in this case) as possible to a network of protected areas (Pressey *et al.*, 1997; Justus & Sarkar, 2002). Hereby, efficiency in area-selection is guaranteed and habitat bias (some habitats are always richer than others) is avoided.

I will introduce two different methods to estimate conservation value: The first method will be a pseudo-complementarity method based on a reference database, calculating an index of site-specific rarity. This method will avoid common problems of habitat bias by correcting richness and rarity for the type of habitat. The second method is a traditional complementarity-based approach (Kirkpatrick, 1983; Justus & Sarkar, 2002) adapted for river conservation. This second method will also account for the effects of catchment degradation by introducing catchment protection rules.

For real-world conservation planning, conservation value alone will not contain enough information to prioritise conservation action. Responding to Myers *et al.* (2000), who mainly prioritise on undisturbed areas, Mace *et al.* (2000) call for conservation planning frameworks that do not include *a priori* knowledge of disturbance. In these frameworks, present condition of the ecosystem will determine the nature of the activity: areas that are still in good condition require protection, restoration activities are appropriate for degraded ecosystems. Terrestrial conservation planners also consider vulnerability – the likelihood that condition will worsen in the future (Bradley & Smith, 2004). This serves as a measure of needed protection effort: highly vulnerable areas need more attention than areas that are not likely to show future degradation through land-use change or other human activities.

I will also add condition and vulnerability as additional properties to the assessment of conservation value to establish a comprehensive framework for river conservation. The first measure of condition is a site-specific metric based on AUSRIVAS (Simpson & Norris, 2000), linking into the pseudo-complementarity method based on a reference database. The second measure of condition, based on stressor gradients (Bailey et al., in press), is more suited for landscape-scale conservation programs, because it only uses GIS variables and does not require site visits. I will also estimate future vulnerability based on present land use, land capability and current level of protection.

To summarise the above, it is the aim of my thesis to establish a complete framework for river conservation, with the following properties (to be discussed in detail in Chapter 2):

1. The estimator of conservation value should be complementarity-based or at least compare the conservation value of a site in the light of its potential.
2. The framework should be geared towards protecting biota, not landscape classes, yet flexible enough to deal with lacking data or the need for other conservation targets (processes, socio-economic factors, etc.)
3. Instead of only estimating conservation value, the framework should also address nature and urgency of conservation action – ideally by considering independent measures of condition and vulnerability
4. The measure of conservation value and the response axes should be geared towards whole-catchment management. This includes protection trade-offs with costs that are associated with whole-catchment protection

This framework will first be established on philosophical grounds. Second, the framework will then be illustrated using macroinvertebrate data in mixed-use environments.

1.3. Structure of the thesis

After this introduction, the second chapter in this thesis will review the relevant literature, including existing methods to estimate conservation value, systematic conservation planning in terrestrial and marine environments and gaps in current freshwater conservation efforts. Chapters three through six are a series of papers that are either published, accepted or ready for submission.

Chapter three '*Biodiversity: Bridging the gap between condition and conservation*' will introduce an alternative measure of freshwater conservation value. This measure is based on a reference condition approach and attempts to achieve a quasi-complementarity estimate by comparing an expected assemblage against a reference database. Also, it will establish a separation of condition and conservation value, which will carry through the following chapters.

Chapter four '*Comparing predictive accuracy in benthic science: Implications for bioassessment and assessment of conservation value*' will compare traditional RIVPACS/AUSRIVAS bioassessment models with ANNA (Assessment by nearest neighbour analysis, Linke et al., 2005). It will highlight problems associated with the approach in Chapter three in real life planning scenarios.

Chapter five '*Irreplaceability in river systems: Towards catchment-based conservation planning*' will introduce a complementarity-based measure of conservation value – used for decades in the terrestrial field – to a riverine environment. Major modifications to the terrestrial system include taxa prediction using GAMs (Generalized additive models, Hastie & Tibshirani, 1999), and a rule to include upstream protection.

Chapter six '*Management options for river conservation planning: Condition and conservation re-visited* ' will translate the measure developed in the previous chapter into a real-life planning scenario. Introducing a vulnerability coefficient *sensu* Wilson *et al.* (2005a), this chapter will set up a framework like Margules & Pressey (2000) in the terrestrial field. However, an additional axis will be added to complete the trinity of aspects to be considered in aquatic conservation planning: Irreplaceability, Vulnerability and Condition.

The synopsis in Chapter seven will then discuss the above approaches and outline future directions in relation to on-ground conservation planning.

Chapter 2. Literature review

The aim of this literature review is to demonstrate the reasons why the two proposed approaches to estimate conservation value were chosen. This review complements the introductions of the following data chapters and highlights background information on theoretical conservation issues that would have been too extensive to include in published articles. Therefore, further references to the scientific literature can be found in each of the data chapters.

2.1. Biodiversity and conservation biology

The assessment and conservation of biodiversity has been one of the most important topics in both academia and natural resource management in recent years. Although the United Nations (UN) Convention on Biological Diversity (UNCED, 1992; UNEP, 1992) is widely recognized, it specifies neither trait, nor method of quantification of biodiversity (Zeide, 1997). The Convention, however, defines biodiversity as "the variability among living organisms from all sources (…) and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (p. 4).

In an aquatic context, Cullen (2003) lists four reasons to maintain biodiversity. First, national (Commonwealth of Australia, 1992; Commonwealth of Australia, 1999) and international (UNCED, 1992, see above) obligations about biodiversity conservation have to be met. Second, protected areas provide a reference condition for comparative assessment of impacted regions (Reynoldson *et al.*, 1997; Bailey *et al.*, 2004) that is needed to underpin ongoing management. Third, protected sites may act as re-colonisation pools for taxa that have been eliminated from other parts of a catchment. Fourth, aquatic species bear intrinsic values (Angermeier, 2000; Sarkar, 2002) and often provide irreplaceable ecosystem services.

Currently, applied biodiversity studies deal with two major issues: biodiversity loss and identification and protection of sites of special significance. Although these streams of biodiversity studies should be integrated in a final assessment, initially they lead to two completely different measures. The first category – biodiversity loss - discussed on local (Crist et al., 2000), national (Smith, 1996; Roper-Lindsay, 2000) and global (Hogg *et al.*, 1996; Williams, 2000) scales, as well as in ecological risk assessment systems (Freedman & Beauchamp, 1998; Kamppinen & Walls, 1999; Reyers & James, 1999) is primarily targeted at areas with a high human impact. Condition assessments in an aquatic setting will be discussed in 2.4.4 (p.22). The outputs from assessments of biodiversity loss include the number of taxa lost, the number of taxa endangered as well as the number of taxa remaining.

The second category deals with description and assessment of biodiversity to estimate conservation value. Methods and existing assessments will be discussed in the following sections 2.2 and 2.3.While a distinction between biodiversity assessment in natural systems and biodiversity assessment in a post-disturbance setting is often ignored, outputs from both should be kept separate (the rationale is explained in detail in Chapter 3), leading to the two approaches trialed in this thesis:

1. The two tiered system – integrating condition and conservation value (see Chapter 3)
2. The integration of a condition axis into established methods in systematic conservation planning (see 2.4.4 and Chapter 6)

2.2. Measures of conservation value

Historically, nature reserves have been selected based on rather unscientific criteria. Pressey *et al.* (2000) state that many areas deemed unsuitable for agriculture and forestry were declared ‘wilderness areas’. The key criterion was therefore plain availability. Other highly objective criteria were mere aesthetic considerations – physical beauty of the landscape (Sarakinos *et al.*, 2001; Neel & Cummings, 2003), leading to inefficient protection of biodiversity. This is discussed in detail by Rodrigues *et al.* (1999) and Scott *et al.* (2001).

Triggered by Ratcliffe's (1971) seminal paper 'Criteria for the selection of nature reserves' – which transformed biological reserve design from an intuitive problem to a technical one (Justus & Sarkar, 2002) - more objective methods to estimate conservation value have been developed. Note that this even pre-dates the actual term 'biodiversity'. The following is only a brief summary about the developments that led to modern assessments of conservation value; an in-depth treatise can be found in Justus & Sarkar (2002).

2.2.1. Richness-based methods

Before the development of complementarity-based approaches for conservation value and reserve design, methods based on taxa richness were the most commonly used approaches. Based on Ratcliffe's (1971) work and various follow-up studies only a few years after (Tubbs & Blackwood, 1971; Tans, 1974; Gehlbach, 1975), selecting the most species-rich areas seemed an intuitive way to prioritise in a subjective fashion. Later, plain richness coefficients were replaced by species-area (SPAR) methods (Simberloff & Abele, 1982; also reviewed by Shafer, 1990).

Richness and SPAR methods are still used today at both regional and global scales. Regional assessments of species richness are more geared to descriptive analyses (Jiguet et al., 2005) or to establish surrogacy relationships (Panzer & Schwartz, 1998) than to measure conservation value. However, on the global scale, richness based assessments (linked with endemism) celebrated a resurgence with the "hotspots" debate (Myers *et al.*, 2000; Hobohm, 2003; Myers & Mittermeier, 2003). In 2000, Myers *et al.* (2000) identified 25 worldwide hotspots as global conservation priorities. This simplified approach attracted a lot of criticism. This criticism was partly for data collection regarded as too unreliable (Brummitt & Lughadha, 2003) and for using an unsuitable estimator that did not include a correction for different species/area coefficients (Ovadia, 2003). Mace *et al.* (2000) added to the criticism, pointing out that any richness-based method cannot deliver optimal conservation results (Kirkpatrick, 1983; see

'Complementarity based methods' below) and that there had been an *a priori* bias towards more or less undisturbed areas.

2.2.2. Diversity/rarity/endemism based methods

While pure richness methods became less popular in the 1970s, these metrics were replaced by indices that accounted for other aspects of community composition at a site. Hobohm (2003) integrates an area-corrected richness method with an area-corrected endemism to assess worldwide conservation value for plants. He argues that purely richness-based methods would miss centres of high endemism (the Hawaii Islands and New Zealand among them). Ricotta (2004) combined relative abundance with Hurlbert’s measure of taxonomic distinctness (Hurlbert, 1971). This integrated measure of conservation value considers higher level taxonomic diversity. For example, a site with a stonefly species, a mayfly species and a dragonfly species is considered as more diverse as a site with three dragonfly species. Higher taxon approaches complement traditional measures of diversity and acts as a coarse measure of genetic diversity (Gaston & Williams, 1993; Gaston, 2000).

The most widely used metrics that do not use complementarity are multi-criteria metrics like the one used by Rey Benayas & de la Montana (2003). Their index of ‘high value vertebrate diversity’ includes a standardized compound measure of richness, rarity and vulnerability. While being more efficient and more philosophically sound than simple richness measures, these composite indices will again not consider what has already been protected and therefore be inefficient (Kirkpatrick, 1983).

2.2.3. Other measures of conservation value

Despite advances in numerical approaches to conservation planning, expert opinion is still used in many planning scenarios (Dinerstein & Wikramanayake, 1993; Mittermeier *et al.*, 1998; Olson & Dinerstein, 1998). While the bias associated with subjective assessment has been criticised by some authors (Kress *et al.*, 1998; Maddock & Samways, 2000), Cowling *et al.* (2003b) point out that expert opinion aids setting protection targets (see below), that are then spatially located by more objective numerical procedures. However, based on a study comparing sole expert opinion and a systematic approach, Cowling *et al.* (2003b) warn that expert opinion by itself is not sufficient to achieve complete protection of taxa or processes.

Genetic diversity is often neglected in assessments of conservation value, despite an explicit mention by the UN biodiversity convention (UNCED, 1992). Ji & Leberg (2002) conducted a GIS-aided assessment of genetic diversity in the Appalachians. Comparing protected area coverage and genetic markers, they identified large gaps needed to ensure genetic diversity. This is in line with the larger theoretical study by Neel & Cummings (2003), who argue that in general assessments of conservation value and reserve design methods lead to a deprivation of genetic diversity if no genetic targets are set. While this is a large field for future research, I only discussed genetic assessments for completeness and –as outlined in the opening section- will not deal with them in the framework of this thesis.

The last group of approaches to prioritise conservation value is based around naturalness. The notion that the state of naturalness itself represents a conservation value was coined in the 70s (Tans, 1974; Gehlbach, 1975; Wright, 1977) - at the same time that Ratcliffe developed the first prioritisation criteria. Despite fierce debate about naturalness as a conservation value-as opposed to a state of condition (Anderson, 1991; Anderson, 1992; Gotmark, 1992; Haila *et al.*, 1997), some conservation scientists are still strong proponents of the concept of naturalness as a conservation imperative (Angermeier, 2000; Machado, 2004). One of the world's only large-scale assessment of river conservation value is based around Australia's 'Wild Rivers', measured by their naturalness (Stein et al., 2002). In my thesis, naturalness will not be used as a conservation imperative, but –in the form of condition- as a driver for the type of conservation action prescribed.

2.3. Complementarity-based approaches

2.3.1. Origins of complementarity-based conservation planning

After ten years of prioritising using simple metrics – richness, diversity, rarity, endemism – Jamie Kirkpatrick, a plant researcher from Tasmania (Australia) questioned whether these single metrics could theoretically lead to the most efficient solution (Kirkpatrick, 1983). In his words (also cited by Justus & Sarkar, 2002):

> "A major drawback of a listing of priority areas on the basis of a single application formula is that there is no guarantee that the priority area second or third on the list might not duplicate the species, communities or habitats that could successfully be preserved in the first priority area".

When he was prioritising for gaps in reserve coverage of plant species, he noticed that many of the high scoring areas were home to the exact same species (Kirkpatrick, 1983; Pressey, 2002). If all of these areas were recommended for prioritisation, conservation effort would be duplicated, while areas with few, yet important species were not assigned high priority. Therefore, Kirkpatrick applied an iterative procedure, marking the highest scoring area as reserved and removing the species covered by this area to adjust the grid valued of the remaining species. This algorithm was called a 'greedy heuristic' – sequentially labelling the highest scoring areas and is still the basis of modern reserve selection techniques.

Ironically, similar reserve selection techniques had been independently discovered by at least two more groups. While Kirkpatrick's data were still unpublished, Ackery & Vane-Wright (1984) worked on a prioritisation strategy for milkweed butterflies (Danainae). They hand-implemented a similar algorithm, based on endemism. When publishing these data seven years later in a peer reviewed journal, Vane-Wright *et al.* (1991) coined the term complementarity. This key principle in modern conservation planning implies that the areas selected in every planning step will complement previously protected areas.

While Kirkpatrick (1983) discovered the principle and Vane-Wright *et al.* (1991) coined the term complementarity, a third group was responsible for formulating the first explicit reserve design algorithm. Working on wetland vegetation, Margules *et al.* (1988) developed a set of selection rules:

1. Select all wetlands with any species that occur only once.

2. Starting with the rarest (i.e. the numerically least frequent species in the data matrix) unrepresented species, select from all wetlands on which it occurs, the wetland contributing the maximum number of additional (i.e. unrepresented) species.

3. Where two or more wetlands contribute an equal number of additional species, select the wetland with the least frequent group of species. The least frequent group was defined as that group having the smallest sum of frequencies of occurrence in the remaining unselected wetlands.

4. Where two or more wetlands contribute an equal number of infrequent species, select the first wetland encountered

With slight modifications, this set of rules became known as the MNP (Margules-Nichols-Pressey) algorithm, or – more scientific – a 'progressive rarity algorithm'.

2.3.2. Comparison of different algorithms

The progressive rarity algorithm is still used in many real-life applications, mainly because of its implementation in leading software packages. Two studies (Csuti *et al.*, 1997; Pressey *et al.*, 1997) compared multiple reserve design algorithms. Hereby, algorithms can be split into two major groups.

The first group, heuristic algorithms – like the three methods described above – use a stepwise selection strategy. This selection strategy can be either based on richness, rarity or summed rarity. Similar to Kirkpatrick (1983), Vane-Wright *et al.* (1991) used a 'greedy richness' algorithm, in which the cell with the most unprotected taxa is selected. The cell and the taxa are then removed and the next cell is selected. This algorithm is still in use in scientific studies (Gladstone & Alexander, 2005; Newbold, 2005) and the WORLDMAP software (Williams, 1999), although both comparisons (Csuti *et al.*, 1997; Pressey *et al.*, 1997) described it as the least efficient approach. The best approach in the comparison was the MNP algorithm mentioned above. Only considering endemic or single rarest taxa at a time, this algorithm assumes that the most rare taxa are the greatest obstacle in achieving complete coverage. Among the heuristic algorithms, the progressive rarity algorithm proved the most efficient in both comparisons. It is currently implemented into two leading software packages- C-Plan (Pressey, 1999) and ResNet (Garson et al., 2002b).

A third heuristic algorithm –termed ‘rarity weight’ or ‘summed rarity’ algorithm (Rebelo & Siegfried, 1990; Rebelo & Siegfried, 1992; Williams *et al.*, 1993b; Kershaw *et al.*, 1994)- combines richness and rarity by summing the inverse frequencies.

$$c = \sum \frac{1}{f}$$

where $c =$ contribution to targets, summed across all taxa in the subcatchment or group of subcatchments, corrected for area

$f =$ frequency of the taxon in the entire dataset

After each step, covered taxa are removed and the frequencies are re-calculated. In the comparison by Csuti *et al.* (1997), this algorithm performed almost as well as the progressive rarity algorithm.

The second large group is formal optimisation algorithms like the branch and bound method (Lawler & Wood, 1966; Nemhauser & Wolsey, 1988; Camm *et al.*, 1996). First used by Underhill (1994), these algorithms try every combination until they find the optimal solution – covering the most taxa or biodiversity using the least area. Unlike heuristics, which only find a near-optimum solution, these algorithms found the best solution in both comparisons (Csuti *et al.*, 1997; Pressey *et al.*, 1997). However, there are two associated problems with optimisation algorithms: First, they use a great amount of computational resources, increasing exponentially with the size of the dataset. Second – even more important in a research study – they lack transparency. In a heuristic algorithm, it remains clear why a particular site or cell is selected and which features are protected by that step. This information is not available in optimisation routines, which prevents researchers or planners from choosing alternatives if a selected cell is not available for protection.

2.3.3. Irreplaceability

While the above heuristic and optimal algorithms deliver a single solution for the species coverage problem, real-life planning scenarios demand alternative solutions in case areas are unavailable or unsuitable for protection or reservation. To gain information about the conservation potential of areas not selected in the minimum set (Ferrier et al., 2000), Pressey *et al.* (1993; 1994) worked towards developing a new indicator of conservation value termed irreplaceability. Pressey *et al.* (1994) defined two operational properties of the new estimators.

1. Irreplaceability reflects the likelihood that an area is required to achieve a set of conservation targets.
2. Irreplaceability reflects the extent that options to achieve a set of targets are reduced by if an area is unavailable.

Over the last 10 years, several approaches have been developed to estimate irreplaceability. Estimators can be classified into three groups: stepwise heuristic estimators (Pressey et al., 1994), statistical estimators (used in C-Plan, Ferrier et al., 2000) and methods that explore multiple possible site combinations (Rebelo & Siegfried, 1992; Possingham *et al.*, 2000; Tsuji

& Tsubaki, 2004). All of these estimators have been applied in many marine and terrestrial studies (see Table 2.1), both in a research context and real life planning scenarios. The table highlights the lack of applications in rivers, as well as a slight bias to statistical and heuristic estimators of irreplaceability.

Table 2.1. Selected previous studies using estimators of irreplaceability.

Authors	Method	Habitat	Target	Location
Roux *et al.* (2002)	Heuristic	Rivers	River types	South Africa
Araujo (1999)	Heuristic	Terrestrial	Plants, reptiles, amphibians	Portugal (USA)
Csuti *et al.* (1997)	Heuristic	Terrestrial	Vertebrates	Oregon (USA)
Williams *et al.* (2003)	Heuristic	Terrestrial	Vertebrates	Guinea/Congo
Wilson *et al.* (2005a)	Heuristic	Terrestrial	Plants	Victoria (Australia)
Woinarski *et al.* (1996)	Heuristic	Terrestrial	Plants, phylogenetic diversity	Northern Territory (Australia)
Tsuji & Tsubaki (2004)	Heuristic/multiple combinations	Terrestrial	Butterflies	Japan
Airame *et al.* (2003)	Multiple Combinations	Marine	Habitat, marine plants, fish and mammals	California
Stewart & Possingham (2005)	Multiple Combinations	Marine	Bioregions, habitat, marine mammals	South Australia
Carroll *et al.* (2003) Noss *et al.* (2002)	Multiple Combinations	Terrestrial	Mammals	Yellowstone (USA)
Gladstone & Alexander (2005)	Statistical	Marine	Molluscs/Fish	South-east Australia
Clark & Slusher (2000)	Statistical	Terrestrial	Plants/Habitat	US Midwest
Cowling *et al.* (2003a) Kerley *et al.* (2003) Rouget *et al.* (2003c) Pressey *et al.* (2003)	Statistical	Terrestrial	Habitat, plants, mammals, process surrogates	South Africa
Ferrier *et al.* (2000) Pressey *et al.* (2000)(2000) Pressey & Taffs (2001) Pressey *et al.* (2004)	Statistical	Terrestrial	Forest types	NSW (Australia)
Warman *et al.* (2004a; 2004b)	Statistical	Terrestrial	Vertebrates	British Columbia, Canada

2.4. Systematic conservation planning - Incorporating condition and vulnerability

2.4.1. Systematic conservation planning

Half a decade after reserve selection algorithms and irreplaceability coefficients as measures of conservation value had been developed, these methods were included into a more comprehensive framework: Systematic conservation planning. In a highly cited article (230 citations up to 2006), Margules and Pressey (2000) introduced a six step framework

1. Compile data on the biodiversity of the planning region

2. Identify conservation goals for the planning region

3. Review existing conservation areas

4. Select additional conservation areas

5. Implement conservation actions

6. Maintain the required values of conservation areas

Hereby, key advances were in steps 2 and 5 - Target setting and prioritizing conservation actions.

2.4.2. Target setting

Step 2 deals with setting proper conservation targets. Targets can be set uniformly ("Every taxon will have to be represented three times") or tailored to meet conservation needs of specific taxa. Furthermore, targets can be set as species directly, but can also be spatial components of the region that serve as surrogates for taxa or ecological processes (Desmet et al., 2002).

Persistence is one of the key properties when setting conservation targets (Gaston et al., 2002), because biodiversity will ultimately succeed at the population level (Hughes et al., 1997). Therefore, in real-world planning exercises critical population numbers will have to be assessed, or at least habitat features have to be selected for persistence (Margules et al., 2002). A good example illustrating the process are the targets set for the Cape Floristic Region in South Africa (Pressey et al., 2003), where three groups of features were targeted: Land types (Cowling & Heijnis, 2001), plant (Pyke et al., 2005) and animal species (Kerley et al., 2003) and biodiversity process (Rouget et al., 2003b). Targets for habitat units were formulated in hectares. For plant species and non-mammal vertebrates presence/absence data was used, but abundance targets for large- and medium sized mammals were set based on population viability estimates. Process targets were again formulated based on landforms that supported critical processes or acted as interfaces. This demonstrates that not only taxa presence/absence is used in modern conservation planning, but also a range of features can be targeted together.

2.4.3. Vulnerability

The second key advance of the integrated framework was the inclusion of a second property – vulnerability. In addition to irreplaceability, vulnerability is a second dimension dictating the urgency of conservation action. When setting conservation priorities in step 4 of the planning framework, irreplaceability of sites, catchments or grid cells is being determined. While the most irreplaceable features still deserve the most conservation effort, vulnerability adds information about the urgency of these efforts. If the present state of a highly irreplaceable an ecosystem is not likely to decline in the near future (Quadrant 3 in Figure 2.1), conservation effort is better concentrated on systems (Quadrant 4) that are vulnerable, thus maximising efficient use of resources (Pressey et al., 2004). This implies that vulnerability in a conservation framework has an inevitable temporal component – degradation or biodiversity loss.

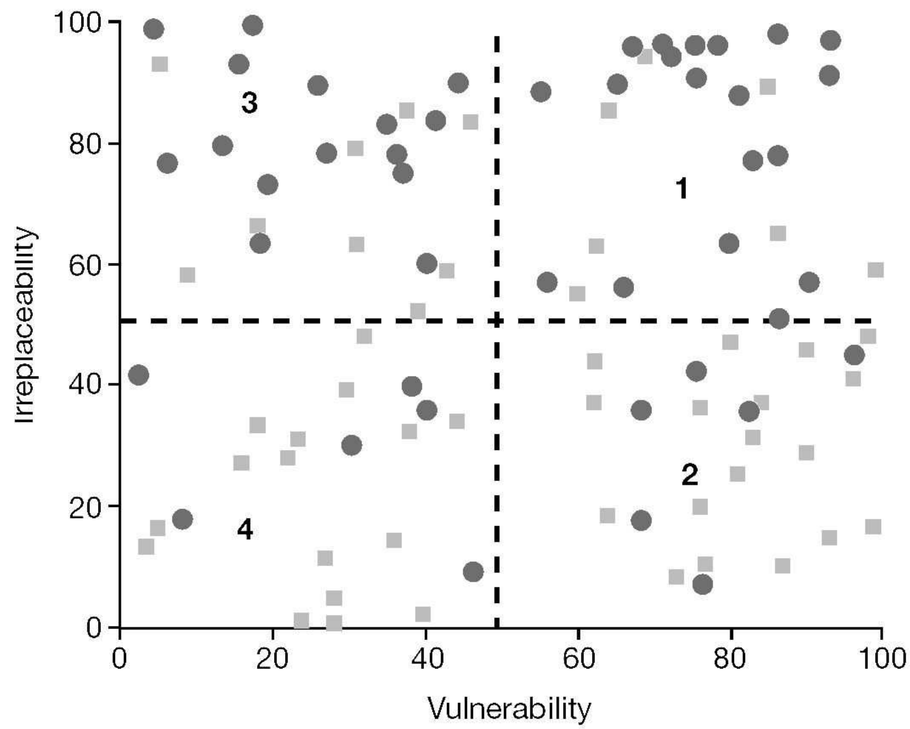

Figure 2.1. 2 dimensions of conservation planning (from Margules & Pressey, 2000).

A more comprehensive treatise of vulnerability in conservation frameworks is given in the introduction of Chapter 6.

2.4.4. Condition as a third proposed axis - *Extending the I-V framework*

Both studies that use ecoregions as potential conservation targets (Pressey *et al.*, 2000; Lombard *et al.*, 2003; Higgins *et al.*, 2005) and studies that use modelled species distributions (Wilson et al., 2005b) often assume that all potential targets are still present in the selected landscape features. While vulnerability in conservation frameworks explicitly includes a change to the current status, an assessment of the present condition is missing in current conservation planning frameworks. In addition to the irreplaceability axis – prescribing the potential effort - and the level of protection needed (as measured by vulnerability), a condition axis would prescribe the nature of the action; protection for land in good condition, or restoration for degraded sites or catchments (Figure 2.2). This is also inline with the split between assessments of condition and conservation value (see above).

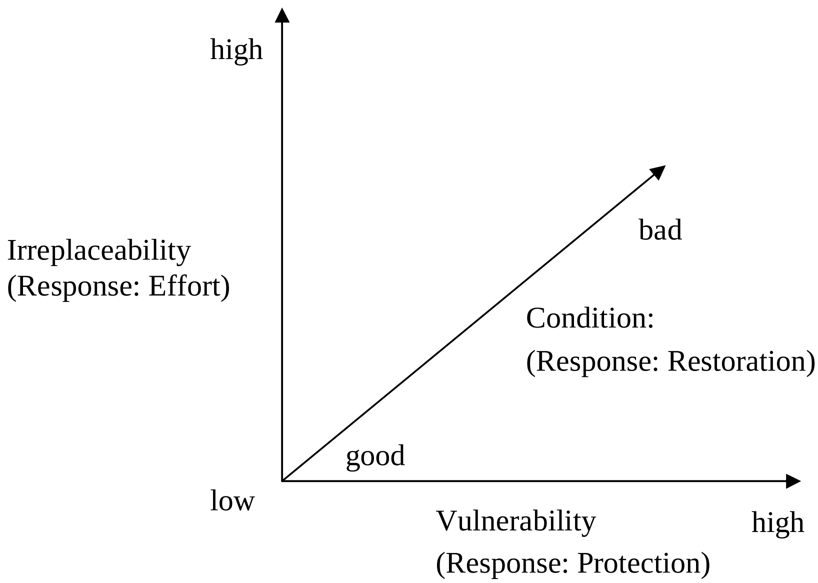

Figure 2.2. The I-V- framework, enhanced by a condition axis

Although the issue of condition has been neglected in terrestrial conservation studies, it has been a main focus of aquatic ecology since the early 1900s. Freshwater condition has mainly be estimated using one of the many site-based approaches, using invertebrates (e.g., Resh *et al.*, 1995; Barbour *et al.*, 2000; Ofenbock *et al.*, 2004), fish (e.g., Ibarra *et al.*, 2003; Bailey & Linke, 2005; Kennard *et al.*, 2005), diatoms (e.g., Chessman *et al.*, 1999; Fore & Grafe, 2002; Bate *et al.*, 2004) or habitat attributes (e.g., Bain & Stevenson, 1999; Muhar *et al.*, 2000; Parsons *et al.*, 2004a).

In recent years, geographic information systems (GIS) have increasingly been used to extrapolate from sampled sites across whole planning regions and link land use and stressor layers to biological condition and biodiversity loss. These assessments - like ARC (Norris *et al.*, 2001; Norris *et al.*, in press), ISC (Ladson et al., 1999) or the stressor-gradient approach

(Bailey et al., in press) - can be used to inform landscape scale planning efforts in conjunction with large-scale irreplaceability and vulnerability assessments.

2.5. Freshwater efforts to determine conservation value – gaps to be addressed

While assessment of river condition has thrived in the last two decades and –some would argue- superseded terrestrial efforts, river conservation science is still lagging compared to terrestrial and marine assessments of conservation value (Daniels *et al.*, 1991; Freitag *et al.*, 1997; Root *et al.*, 2003) and systematic conservation planning (Margules & Pressey, 2000; Abell, 2002; Sarkar *et al.*, 2002; Cowling *et al.*, 2003a; Pierce *et al.*, 2005). Reserves are usually targeted towards terrestrial features (Lake, 1980; Maitland, 1985; Nilsson & Gotmark, 1992; Skelton *et al.*, 1995), with a few notable exceptions (Bayley *et al.*, 1991; Saunders *et al.*, 2002).

When prioritising for freshwater conservation value, most studies are either geared towards single species/taxonomic groups or use index-based measures. Most of the index-based measures are still richness metrics (Clavero et al., 2004), measures of rarity (Filipe et al., 2004) or compound richness/rarity/diversity indices (Williams et al., 2004). A more sophisticated approach used an Index of Centers of Density (Angermeier & Winston, 1997), which identifies regionally rare fish taxa and assesses relevance of sites as sources for potential recolonisation. Nevertheless, all of these methods are inefficient by Kirkpatrick's (1983) criteria: None of them considers complementarity, making all of the approaches less efficient than modern terrestrial and marine conservation planning systems.

The British SERCON (System for Evaluating Rivers for Conservation; Boon *et al.*, 1998; Raven *et al.*, 1998; Boon, 2000; Boon *et al.*, 2002) is the most sophisticated multimetric index for conservation planning and one of the most comprehensive tools. SERCON considers physical diversity, naturalness and representativeness in addition to richness, rarity and special features. Furthermore, 11 different classes of impacts are assigned to a reach. Despite a complex weighting system, the final index is summarized to a single metric, resulting in a loss of information, especially when compared to the separate axes of the I-V-framework. Despite considering catchment effects in some of the indices, SERCON also does not prescribe catchment-scale conservation measures.

In the last five years, the first efforts resembling systematic conservation planning for rivers surfaced. Studies in the both the USA (Smith *et al.*, 2002; Weitzell *et al.*, 2003; Higgins *et al.*, 2005) and Australia (Fitzsimons & Robertson, 2005) called to establish aquatic bioregionalisations. Bioregions are used as representation targets in a coarse-filter CAR (comprehensive, adequate and representative) approach, similar to the one used by Lombard *et al.* (2003) and Pressey (2000). However, the efficiency of broad classifications has been debated heatedly (Araujo *et al.*, 2001; Brooks *et al.*, 2004a; Higgins *et al.*, 2004). The consensus is that if environmental attributes are not stratified rigorously, species might not have are better representation than in a random allocation of planning units (Lombard et al., 2003).

In the Cape Floristic Region study – one of the few studies that studied surrogates in a systematic way – plant species were represented reasonably well by land classes (59%-79% species covered, depending on the method). Because of the weak representation of vertebrates (only 6%-35% species covered), the authors recommended to at least run a mixed algorithm with both species and land classes. In this thesis, I will therefore attempt to directly relate habitat features to species and use this relationship to set conservation targets.

The only real application of an algorithm based planning method in a freshwater setting was a planning exercise on physical data in South Africa's Greater Addo Elephant National Park. Lacking good biological data, Roux *et al.* (2002) used river flow patterns, geomorphological zones and ecosystem processes as surrogates to use in C-Plan. Also lacking a proper framework for upstream catchment protection, they only included entire river lengths as potential conservation zones. This is consistent with a notion many freshwater conservation scientists bear: upstream areas must be considered when estimating conservation value or developing a system of protected areas (Moyle & Sato, 1991; Puth & Wilson, 2001; Crivelli, 2002; Collares-Pereira & Cowx, 2004). However, a more flexible approach is desirable because protection of entire rivers might be difficult to achieve.

A new framework for freshwater conservation planning – the aim of this thesis – should address the gaps outlined above and have the following properties:

1. The estimator of conservation value should be complementarity based or at least compare the conservation value of a site in the light of its potential.
2. The framework should be geared towards protecting biota, not landscape classes, yet flexible enough to deal with lacking data or the need for other conservation targets (processes, socio-economic factors…)
3. Instead of only estimating conservation value, the framework should also address nature and urgency of conservation action – ideally by considering independent measures of condition and vulnerability
4. The measure of conservation value and the response axes should be geared towards whole-catchment management. This includes protection trade-offs with costs that are associated with whole-catchment protection

Chapter 3. Biodiversity: Bridging the gap between condition and conservation

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3.1. Abstract

The aim of this study is to create a two-tiered assessment combining restoration and conservation, both needed for biodiversity management. The first tier of this approach assesses the condition of a site using a standard bioassessment method, AUSRIVAS, to determine whether significant loss of biodiversity has occurred because of human activity. The second tier assesses the conservation value of sites that were determined to be unimpacted in the first step against a reference database. This ensures maximum complementarity without having to set *a priori* target areas. Using the reference database, we assign site-specific and comparable coefficients for both restoration (Observed/Expected taxa with >50% probability of occurrence) and conservation values (O/E taxa with <50%, rare taxa). In a trial on 75 sites on rivers around Sydney, NSW, Australia we were able to identify three regions: 1) an area that may need restoration; 2) an area that had a high conservation value and; 3) a region that was identified as having significant biodiversity loss but with high potential to respond to rehabilitation and become a biodiversity hotspot. These examples highlight the use of the new framework as a comprehensive system for biodiversity assessment

3.2. Introduction

The assessment and conservation of biodiversity has been one of the most important topics in both academia and natural resource management in recent years. Although the United Nations (UN) Convention on Biological Diversity (UNCED, 1992; UNEP, 1992) is widely recognized, it specifies neither trait, nor method of quantification of biodiversity (Zeide, 1997). The Convention, however, defines biodiversity as "the variability among living organisms from all sources (…) and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". This implies that there should be different measurement and management tools for each scale (Bass *et al.*, 1998; Lister, 1998). The current debate on biodiversity can be divided into academic versus applied goals (Srivastava, 2002). The academic side of biodiversity research mainly focuses on the links between biodiversity and ecosystem function/stability, using process-based analysis to set conservation strategies (Bengtsson, 1998; Lister, 1998; Schwartz *et al.*, 2000). Applied studies, geared towards managing biodiversity, take a more reductionist point of view (Lister, 1998), quantifying species and populations.

Applied biodiversity approaches can be separated into two major groups. Most UN efforts at present are focussed on status, trends and causes of biodiversity loss (UNEP, 2003). The most commonly used framework for these restoration efforts is the Pressure-State-Response framework, developed by the OECD and described in Cairns & Pratt (1995). Species loss has been discussed on local (Crist et al., 2000), national (Smith, 1996; Roper-Lindsay, 2000) and global (Hogg *et al.*, 1996; Williams, 2000) scales. These studies are also the basis for ecological risk assessment systems (Freedman & Beauchamp, 1998; Kamppinen & Walls, 1999; Reyers & James, 1999), geared towards applied ecosystem management. This branch of applied biodiversity research is mainly focussed on areas with a high human influence that puts ecosystems at risk.

The second branch of applied biodiversity science includes conservation studies. Instead of discussing loss of biodiversity, publications by Freitag *et al.* (1998), Margules *et al.* (2002) and Myers *et al.* (2000) focus on the conservation of sites of special significance, mainly in areas that are not affected by human activities. Many of these studies deal with the problem of how to identify and protect areas of high conservation value, based on species richness, and endemism or rarity (Mittermeier *et al.*, 1998; Noss, 2000). While conservation studies are often confined to areas with low pressure from human activities (Pressey *et al.*, 2000; Simonson *et al.*, 2001; Desmet *et al.*, 2002), Mace *et al.* (2000) call for a system that does not require *a priori* selection of target areas and that operates on a smaller scale than national and global assessments. In our view, a strategy that integrates the assessment of loss of biodiversity, combined with the selection of patches of special significance (or high biodiversity) would unify both branches of applied biodiversity research and create a powerful management tool.

Another extensively discussed topic is the technical aspects of biodiversity assessment. Although precision is always a pre-requisite for scientific studies, cost effectiveness is a key issue for diversity surveys (Danielsen *et al.*, 2000; Gioia & Pigott, 2000). The concept of surrogates in biodiversity assessments has been widely accepted in the recent years. Surrogates are taxonomic groups that indicate the overall biodiversity at a surveyed site. Although multi-taxa studies are preferable, researchers recommend invertebrates as the group that will represent up to 90% of the genetic variation (Duelli, 1997). In biomonitoring surveys of aquatic systems, benthic invertebrates have played a key role for years, because they are ubiquitous and diverse (Rosenberg & Resh, 1993), relatively inexpensive to sample and many laboratories have a good working knowledge of their taxonomy (Hellawell, 1986). In this paper, we trial the use of benthic macroinvertebrates as surrogates for aquatic biodiversity in inland rivers.

The aim of this study is the development of a two-tiered approach for applied biodiversity studies. Our model system will be rivers in the Sydney water supply catchments, Australia; an area with patchy land use ranging from national parks and agricultural and urban systems. The first step will be the identification of areas with significant biodiversity loss using AUSRIVAS (Australian River Assessment Scheme, Simpson & Norris, 2000), a RIVPACS-style method (Wright et al., 1993a) for assessing aquatic ecosystem health. After sieving out river reaches that suffered from significant biodiversity loss, we will identify areas of special conservation interest using the AUSRIVAS reference database, to determine a site-specific index of conservation value (areas with higher than expected richness). This two-tiered approach will merge both branches of applied biodiversity assessment and thus meet an important need for managing both condition and conservation.

3.3. Methods

3.3.1. Study area and sampling methods

The catchments that supply water to Sydney cover about 16000 km^2 in south-east New South Wales, Australia. The major land-use types are:

- protected areas (native vegetation, forested mountain areas, national parks, nature reserves): 49 %
- agricultural/forested sites (mainly sheep and cattle grazing): 49.5%
- urban areas: 1.5 %

Thirty nine sites little affected by human activities (mostly in protected areas) were chosen for building a reference database. Some sites were also chosen from catchments adjacent to those used for water supply. Test sites chosen for assessment included 11 from protected areas, 15 in agricultural areas and 10 urban.

Macroinvertebrates were collected from edge habitats (slow flowing, with structure provided by aquatic or overhanging vegetation, tree roots, large woody debris or bank undercutting), using a kick-net 350 mm wide with 500 µm mesh for a total transect of 10 m, as described in Turak *et al.* (1999). A composite sample from each habitat at a site in proportion to its representation, analogous to Wright *et al.* (1993a), was collected for the test sites to determine whether the edge samples were appropriate surrogates for the biodiversity of the site. Invertebrates were picked from the whole samples using a modified New South Wales EPA (Environmental Protection Agency) live-pick method (Turak et al., 1999) with a minimum of 200 animals retrieved. To ensure a rapid assessment, macroinvertebrates were only identified to family, apart from the orders Plecoptera, Odonata, Ephemeroptera and Trichoptera, which were identified to species.

3.3.2. Assessment of condition and biodiversity loss

Site condition and possible loss of biodiversity (Tier 1, Figure 3.1) were assessed using AUSRIVAS, the standard method for river health assessment in Australia (Simpson & Norris, 2000). AUSRIVAS predicts the probability of taxa occurring at a test site from a reference database of undisturbed sites (Turak *et al.*, 1999; Simpson & Norris, 2000) by matching the environmental characteristics. Only taxa with a probability of occurrence higher than 50% are considered in the final assessment. By summing both probabilities of occurrence and the number of taxa collected, the coefficient of Observed/Expected (OE50) can be calculated (Table 3.1).

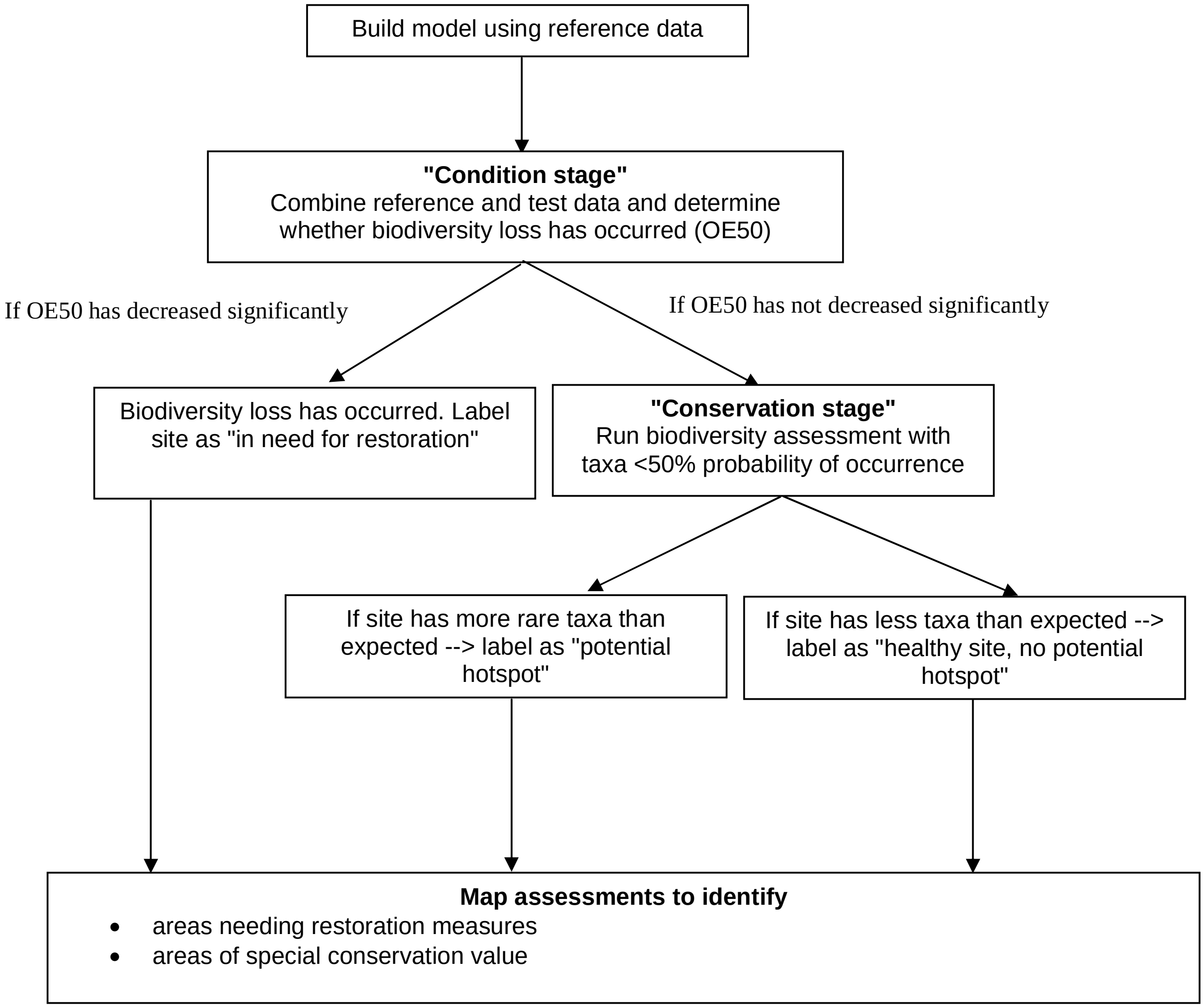

Figure 3.1. Flowchart for a two-tiered approach, integrating assessment of condition and conservation value

Table 3.1. **Coefficients for biodiversity loss (OE50) using taxa with a probability of occurrence >50%**

	Observed (Presence)	Expected (Probability)	
Taxon 1	1	0.5	
Taxon 2	1	0.7	
Taxon 3	1	0.6	
Taxon 4		0.8	
Sum	**3**	**2.9**	**O/E=1.03**

The OE50 is a site-specific coefficient measuring loss of taxa. Only the more common taxa are considered (>50% probability of occurrence) and this makes the predictive model less prone to error by focussing only on the taxa "that should be there" to assess whether the normal state is present or biodiversity has been lost. To ensure a Type I error of 10%, the 10th percentile of the distribution of reference sites will be used as the cut-off for a significant loss of biodiversity. For example, if the cut-off is 0.8 and only 7 out of 10 expected taxa are found at the test site, a significant loss of biodiversity is detected. If a significant loss is detected, the site will not be considered for the second stage, the assessment of conservation value (see Figure 3.1).

3.3.3. Assessment of conservation value

Conservation value is focussed on the rare taxa (Tier 2, Figure 3.1) in contrast to the assessment of condition, which is based around the common taxa at a site (defined by the probability of occurrence). Although "rare species" is a high profile term in biodiversity literature, there are few working definitions or scientific criteria for making a determination. In this study we define a taxon that has a < 50% chance of occurrence at a site as rare (in future studies, this could be shifted to a lower boundary). When using a reference database, this site-specific assessment of regional rareness, will define "rare taxa" based on the operational management unit, which fulfils the requirements by Mace *et al.* (2000). The conservation coefficient OE(BIODIV) is calculated analogous to the OE50, but uses only taxa with <50% probability of occurring at a site (Table 3.2). While only 20 to 30 taxa might be included in the OE50, the whole remaining taxa in the reference database will be included. If the 250, or so remaining taxa in the database have an average probability of occurrence of 0.08, the expected number of rare taxa will be 250 x 0.08 =20. If 20 taxa were found at this site, the site would be "as expected", with an OE(BIODIV) of 1. If the observed number was greater than the expected, for example when taxa are found that do not exist in the reference collection or that are not expected in the particular area, the coefficient increases, labelling the site as taxonomically richer than expected or a possible conservation ‘hotspot’.

Table 3.2. Coefficients for conservation value OE(BIODIV) using taxa with a probability of occurrence <50%

	Observed (Presence)	Expected (Probability)	
Taxon 5	1	0.01	
Taxon 6		0.3	
Taxon 7		0.2	
Taxon 8		0.1	
Taxon 9	1	0.2	
Taxon 10		0.4	
Taxon 11		0.3	
Taxon 12	1	0.25	
Sum	3	2.46	O/E=1.21

3.3.4. Spatial analysis

To identify areas where restoration is needed or parts of the catchment that are of high conservation value, we mapped both condition and conservation assessment using ARCView 3.2 (ESRI, 1998).

3.4. Results

The edge habitat can be seen as a good surrogate for the entire macroinvertebrate diversity at a site. The r^2 between species richness in the edge habitat and composite from all habitats was 0.5 ($p<0.001$). At 1.3 species/family at sites on average, the number of species was correlated to the number of families at $r^2=0.83$ ($p<0.001$), clearly indicating that effective assessment could be achieved with identifications to only family level.

The species level AUSRIVAS model developed using the 39 reference sites was acceptable. The correlation of observed to expected taxa in the reference sites (ideally 1) was $r^2=0.48$.

The 10th percentile cut-off of OE50 scores to determine significant loss of biodiversity was 0.7

Of the 75 assessed sites, 29 sites failed the condition assessment (OE50<0.7), including 4 reference sites that necessarily failed by definition. Only one of the sites with urban influence and only three agricultural sites passed the assessment. Seven of the sites in near natural condition had loss of biodiversity. The conservation assessment was run on the remaining 44 sites and their OE(BIODIV) scores calculated (Figure 3.2). The distribution is centred around 1 and quite wide (0.5-1.6, Figure 3.2).

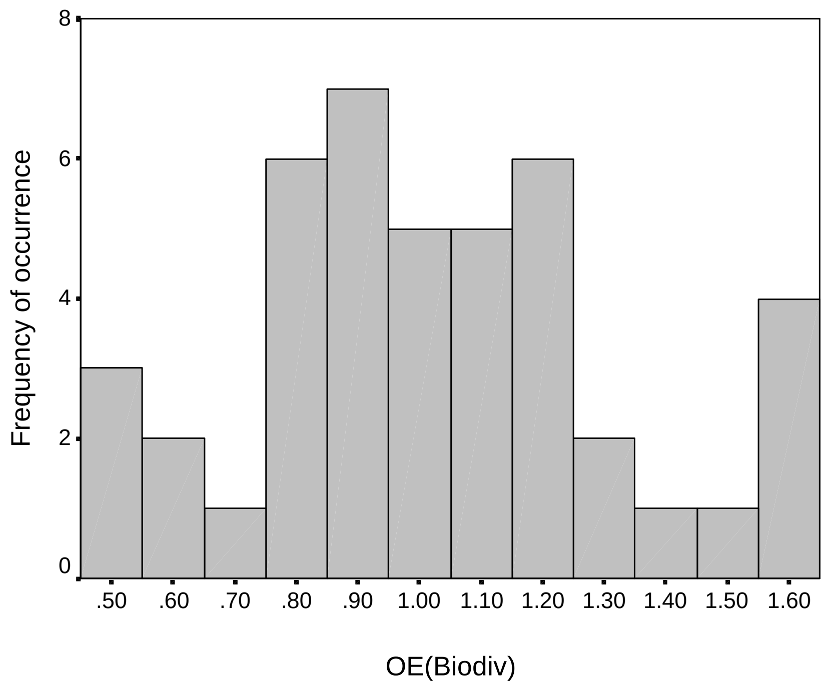

Figure 3.2. Histogram of OE(BIODIV) values for unimpacted sites in catchments of the Sydney region

Mapping the scores revealed that the Clyde river in the south-west of the study area (Figure 3.3) had several adjacent sites with high OE(BIODIV) scores, indicating many rare taxa and a high potential conservation value. Hardly any sites in the mostly agricultural central Shoalhaven valley passed the condition assessment. This highlights a significant loss of biodiversity and the need for restoration measures. The upper Cox's River in the Blue Mountains has many sites affected by human activities on the main stem, but high OE(BIODIV) values indicate that this catchment could be a potential biodiversity hotspot if provided adequate protection (Figure 3.3).

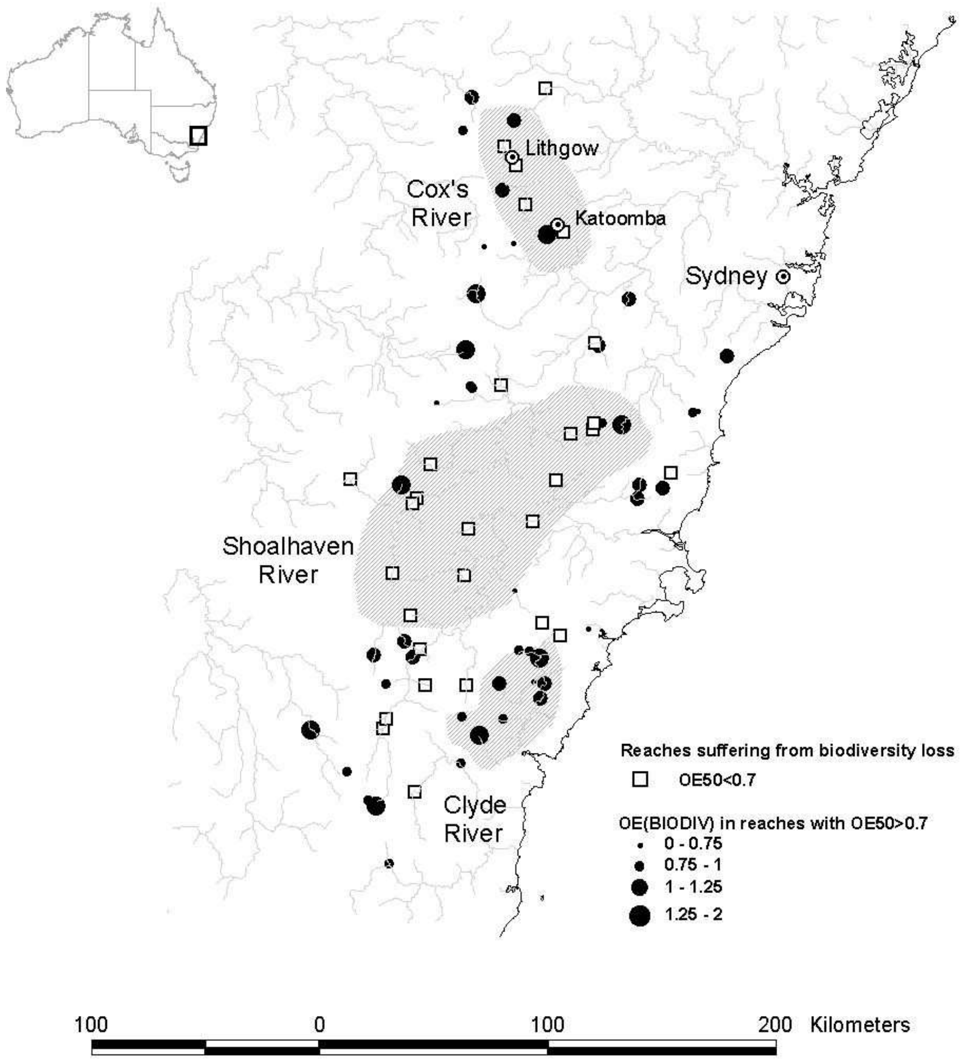

Figure 3.3. Biodiversity condition and conservation potential for rivers in catchments of the Sydney region.

3.5. Discussion

Our study demonstrates a potential method for addressing some frequently debated issues in applied biodiversity research. First, we attempted to integrate an assessment of biodiversity loss and the evaluation of conservation value. Second, we introduce a site-specific and comparable scoring algorithm to set conservation priorities, enabling a comparative evaluation at the scale of a management unit. Third, our study also tested the validity of rapid assessment using family rather than species level taxonomy and a single habitat rather than composite habitats.

Although we used mostly species level data for further analysis, the high correlation of family versus species richness and the low ratio of species/family at a site (1.25) suggests that family data would be an appropriate surrogate in future studies. Despite concerns about the use of family data in stream ecology (Lenat & Resh, 2001), there is evidence for the validity of lower taxonomic resolutions, both on an empirical (Marchant, 1990; Bailey *et al.*, 2001) and a functional level (Thompson & Townsend, 2000). Error that may be introduced by using families rather than species may be more than made up for by using saved resources to sample additional sites that would greatly strengthen spatial assessment (Figure 3.3). We are currently designing a future study that includes aquatic plants, diatoms and fish to test whether macroinvertebrates are an appropriate surrogate for biodiversity as suggested by Duelli (1997) for terrestrial habitats.

Another cost-reducing factor would be the concept of a "representative habitat". The high correlation of taxonomic richness in edge and composite habitat in the test dataset indicates that one habitat is sufficient to characterise the macroinvertebrate diversity in a study site. If it proves representative in further studies, a protocol using limited taxonomic groups, a relatively low taxonomic resolution (i.e. family or genus level) and a representative habitat would lead to a cost-effective rapid biodiversity assessment, analogous to the existing rapid bioassessments in the river health literature (Plafkin *et al.*, 1990; Barbour *et al.*, 1992). A rapid biodiversity assessment covering both loss and conservation potential would satisfy calls for cost-effectiveness (Danielsen *et al.*, 2000; Gioia & Pigott, 2000) and would enable higher spatial resolution of the surveys meeting management needs on both counts.

The use of a reference condition approach (Reynoldson et al., 1997) introduces the ability to making objective comparisons in biodiversity assessments. The common reserve selection algorithms (Freitag *et al.*, 1997; Margules *et al.*, 2002) all include rarity of taxa as a value, an issue also discussed by Noss (2000) and Sarkar *et al.* (2002). A predictive approach from a reference database adds an objective, a-priori defined measure of rarity within the study region, which gives more weight to unusual or endemic taxa. OE(BIODIV) (Table 3.2) is a site-specific measure of the richness of taxa that are rare within the study region. The approach is more than a mere stock take of taxa richness. It compares the observed occurrence of rare taxa to the potential biodiversity of the site given its location and characteristics.

As a selection algorithm, the calculation of OE(BIODIV) is an attempt to resolve the problem of "complementarity" (Faith & Walker, 2002; Justus & Sarkar, 2002). The principle of "complementarity" in conservation theory looks for sites that add as many under-represented surrogates (taxa in this case) as possible (Pressey *et al.*, 1993; Faith & Walker, 2002; Sarkar *et al.*, 2002). Using a reference database that mainly contains protected sites as the benchmark will ensure an assessment based on complementarity. Sampling of taxa that are not ubiquitous in the reference database or not expected in a certain sub-catchment will increase the OE(BIODIV) and therefore the complementarity value of the site. This ensures the identification of surrogates in a data-driven and repeatable way (Desmet et al., 2002).

Following the global prioritisation of biodiversity hotspots (Myers et al., 2000), Mace *et al.* (2000) called for techniques in biodiversity assessment that would be applicable to a range of spatial scales, especially smaller scales that would have more relevance to management units. A reference-based approach would be applicable to all scales, yet might primarily be applied within management units. Reference databases are usually compiled by the same authorities that are responsible for future conservation and restoration planning.

For river bioassessment, reference databases have been established in many parts of the world. Apart from RIVPACS/AUSRIVAS predictive model approaches in Australia (Simpson & Norris, 2000), Great Britain (Moss et al., 1987), Canada (Reynoldson *et al.* 1997), Spain (Alba-Tercedor & Pujante, 2000) and Indonesia (Sudaryanti et al., 2001), other approaches using reference data have been applied in the Yukon territory (Bailey et al., 1998) as well as highly urbanized areas of Canada (Linke et al., 1999) and Europe (Wimmer et al., 2000). Reference databases also exist in most areas of the USA, either used for RIVPACS models (Hawkins et al., 2000) or multimetric assessment (Karr & Chu, 1999). These databases could be readily used or enhanced. Depending on the spatial scale and resolution, as well as the taxonomic level, new databases can be relatively inexpensive to build.

Apart from adding innovative approaches to conservation planning, the main aim of this study was to integrate the assessment of condition and conservation value (Figure 3.3). The two-tiered approach proposed in this contribution satisfies the criteria specified by Mace *et al.* (2000). In their critique of Myers *et al.* (2000) they called for a system that has no a-priori selection of target areas for biodiversity studies. Confining conservation efforts to natural areas discards the information of areas with mixed land use and does not maximise complementarity. The case studies in the three sub-catchments used in our study illustrate the flow of a possible management system. The Clyde catchment in the southeast of Figure 3.3 is an area largely in reference condition. Almost all of the sites score above average in the OE (BIODIV), some as high as 1.6, indicating that the macroinvertebrate composition at these sites is unique and has high conservation value. The condition of the middle Shoalhaven catchment suggests that this area suffered from a severe loss of biodiversity, highlighting the need for restoration measures. At this stage, an assessment of the conservation value of the middle Shoalhaven catchment cannot be provided, because almost all of the sites were assessed as suffering biodiversity loss. The upper Cox's River catchment demonstrates the real value of a two-tiered approach: Although the sites on the main stem, downstream from the urban centres of Lithgow and Katoomba have suffered a loss of species richness, the unimpacted tributaries have a high complementarity value. This suggests that with appropriate catchment management, the upper Cox's River could be restored to be a major hotspot for macroinvertebrate diversity and worthy of conservation. The area is taxonomically rich and can readily provide colonists to rehabilitated areas also indicating that it is likely to be responsive to management intervention. This conclusion could not have been reached with a traditional approach that only targets untouched areas for conservation management and highlights the need to integrate condition and conservation.

This study was a pilot project, intended to demonstrate the philosophy of the approach. Issues of taxonomic and spatial resolution, as well as adequate surrogacy of macroinvertebrates will be examined in a follow-up project. It also seems desirable to include socio-economic factors into the decision tree as demonstrated in Faith & Walker (2002). Overall, the reference condition guided, two-tiered approach, addresses many issues raised in the recent literature. It is cost-effective (Danielsen *et al.*, 2000; Gioia & Pigott, 2000), data-driven and repeatable (Desmet et al., 2002) and adds a quantitative, comparative approach to biodiversity assessment (Duelli, 1997). Integrating condition and conservation avoids the problem of information loss by setting *a priori* target areas (Mace et al., 2000) and maximises complementarity. Simple outputs that can be applied at large spatial scales will aid both restoration decisions and identification of conservation priorities and thus will provide a more comprehensive tool for biodiversity assessment that meets an urgent need for managers.

Chapter 4. Accuracy of predictive models at the taxon level: implications for bioassessment programs and biodiversity assessment

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Linke, S., Hawkins, C.P., Bailey, R.C., Yuan L.L. Accuracy of predictive models at the taxon level: implications for bioassessment programs and biodiversity assessment

4.1. Abstract

In this study, we examined the accuracy of single taxa predictions by multivariate prediction methods to assess 1) models for bioassessment and 2) models for biodiversity assessment (*sensu* Linke & Norris, 2000). Usually only the final metrics derived from the predicted taxa in such models are tested. Using three datasets from the USA and Canada, we found that the classification step in RIVPACS (River Invertebrate Prediction and Classification System) models influences the nature of the taxa that are predicted. Taxa that have a higher indicator value for one of the RIVPACS groups are more likely to be predicted correctly at a site. Overall, ANNA (Assessment by Nearest Neighbour Analysis) models that do not include the RIVPACS classification step succeed in predicting a wider variety of taxa and more individual occurrences. Our findings have consequences for two applications of these models: Firstly, bioassessment models that predict an increased number of taxa are more sensitive than models predicting fewer taxa. Our recommendation is to use single taxa predictions as a standard indicator of model quality in future studies. Secondly, when used for biodiversity assessment *sensu* Linke & Norris (2003), an accurate prediction is essential, as otherwise common taxa will be declared rare due to model error. The results from this study indicate that often O/E(BIODIV) is not applicable because not enough natural variation can be explained by the models.

4.2. Introduction

Over the last twenty years, predictive modelling approaches have become essential for bioassessment programs worldwide (Hawkins *et al.*, 2000; Mazor *et al.*, 2006). The rationale of predictive modelling in bioassessment is to account for natural variation in biological communities by predicting the state of ecological assemblages under reference conditions (Reynoldson *et al.*, 1997; Bailey *et al.*, 2004). This modeled state is then compared to an observed assemblage at a test site. While predictive models can be built based on metrics such as richness, diversity measures or biotic indices (Bailey *et al.*, 1998; Linke *et al.*, 1999; Bailey *et al.*, in press) most predictive modelling approaches predict whole assemblages (some are consecutively reduced back to indices).

These assemblage-based models have been built with a range of techniques. Olden (2003) used artificial neural networks to predict fish assemblages, Chessman (1999) built distance-dissimilarity models for macroinvertebrates (akin to Faith, 2003; Faith *et al.*, 2004) and Chessman and Royal's (2004) filters approach uses ecological niche theory. However, the most prevalent strategies to predict ecological assemblages are the RIVPACS (River Invertebrate Prediction and Classification System) family of models. This technique – widely used in the UK (Wright *et al.*, 1993a; Moss *et al.*, 1999; Clarke *et al.*, 2003), the USA (Hawkins *et al.*, 2000; Ostermiller & Hawkins, 2004) and Australia (Simpson & Norris, 2000; Barton & Metzeling, 2004; Hose *et al.*, 2004) first classifies assemblages into groups with a clustering algorithm. A discriminant function analysis (DFA) is then carried out to predict the membership of a test site to establish the site-specific reference condition. The summed probabilities of occurrence among predicted taxa are then compared to the presence or absence of the corresponding taxa, and the ratio of observed/expected taxa used to measure departure from reference condition.

The RIVPACS literature in the 1990s and early 2000 was mainly restricted to studies about the development of a system for assessing freshwater ecosystems using macroinvertebrates (Wright *et al.*, 1993a; Wright, 1995; Wright *et al.*, 1998) and its statistical properties (Clarke *et al.*, 1996; Moss *et al.*, 1999) as well as its spread to Australia (Smith *et al.*, 1999; Turak *et al.*, 1999; Simpson & Norris, 2000), Canada (Reynoldson *et al.*, 1995; Reynoldson, 2001) and the USA (Hawkins & Norris, 2000; Hawkins & Vinson, 2000). Recently, researchers mainly focussed on expanding the application of RIVPACS to other taxonomic groups including fish (Joy & Death, 2002; Hawkins, in press; Kennard *et al.*, in press) and periphyton (Mazor et al., 2006), but also on validation and sensitivity analysis.

Validation and calibration of RIVPACS models so far can be divided into two groups. The first group of validations are related to sensitivity: a group of test sites with real or simulated impairment is assessed. The success of a RIVPACS model is measured by the rate of detection of these impacts. Examples include:

- Ostermiller & Hawkins (2004) and Linke *et al.* (1999) define sensitivity as the rate of failing any sites that do not qualify as reference sites due to *a priori* defined disturbance.
- Mazor *et al.* (2006) test against pre-determined stressor groups (agriculture, mining, pulp mills, urban and multi-stressor).
- Mazor *et al.* (2006) also simulated impacts, in a similar approach to Bailey *et al.* (2004) and later Cao & Hawkins (2005), who used a randomised approach to remove sensitive taxa from reference sites
- Sloane and Norris (2003) and Linke *et al.* (2005) used a trace metal gradient to demonstrate a dose-response relationship.

The second group of validations looks at the specificity of the models by evaluating predictions at reference sites. The expected assemblage at a reference site should be identical to the actual sampled assemblage. Any deviation from this can be attributed to model error. The most common way to evaluate models is to calculate the standard deviation of RIVPACS' observed/expected (O/E) score (SD O/E). A wider standard deviation around the optimal mean of 1 indicates a higher error, while a smaller value signals a better prediction. Van Sickle *et al.* (2006) combine bias and precision by using the RMSE (root mean square error). While all of these evaluate the indices derived from predicted and observed data, no study so far has examined the predictive accuracy of RIVPACS models at the taxon level.

Taxonomic accuracy is relevant in two circumstances. First, in standard use of RIVPACS it determines sensitivity of the assessment. Most bioassessment programs worldwide only use taxa that are predicted at a probability of 0.5, which delivers a more robust assessment (Hawkins *et al.*, 2000; Simpson & Norris, 2000) by reducing unexplained variation. Only the taxa that have a high probability of being present count in the assessment. A taxon that is falsely predicted to be present will affect model assessment by lowering the O/E score for the site and therefore affecting SD(O/E) or the RMSE. However, if a taxon is not predicted but present, this will not directly affect the common measures of determining model quality as the presence will not be counted.

However, these non-predictions affect model sensitivity. Let us imagine two hypothetical RIVPACS models. Model I only predicts four taxa (Taxon A,B,C,D), while Model II predicts ten taxa (Taxa A-D plus E-J). If human disturbance caused a local extinction of taxa E-J, this would remain undetected in Model I, but get picked up by Model II, causing a drop in O/E. Therefore a model that accurately predicts more taxa will be a better model, as it is sensitive to the loss of more taxa.

The second reason to evaluate taxonomic accuracy is the application of RIVPACS for conservation purposes. Wright *et al.* (1993b) were the first to use RIVPACS models with the objective of classifying sites with higher than expected taxa richness. While Wright *et al.* (1993b) defined high conservation value sites by RIVPACS O/E, Linke & Norris (2003) modified the system by only considering taxa predicted at probabilities below 0.5. Their index - termed OE(BIODIV) or $OE_{below50}$ – was calculated analogous to the O/E, but only considered the 'rare' taxa, as defined by $p<0.5$. A site that had more occurrences of 'unexpected' taxa would score higher, especially if the probabilities of occurrence were lower.

This system of site-specific rarity – defined by Linke & Norris (2003) as 'unexpectedness' - can only work if the model explains a maximum of the natural variation. If taxa are not predicted because they could not be expected at the site even with a perfect model, they are a valid addition to the index of site-specific rarity. However, if a taxon is not predicted because the model is error prone or simply not powerful enough, the taxon is wrongly classified as rare.

In this study, we assessed the ability of RIVPACS models to predict single taxa and assess the influence of the classification step. The quantification of classification's influence on model output will be performed by comparing RIVPACS models to a more recent technique (ANNA, Linke et al., 2005), which operates on a nearest neighbour method without a classification step. Based on the success of predicting taxa in three geographically distinct bioassessment studies, we will examine how predictive accuracy influences bioassessment and evaluation of sites with respect to conservation value. We will also compare RIVPACS and ANNA models in respect to O/E BIODIV and the influence of taxa prediction on assigning site-specific rarity.

4.3. Methods

4.3.1. Data

We built predictive models using data from three studies of wadeable streams in 1) North Carolina, 2) the Skeena region of north-western British Columbia, and 3) the Yukon River Basin. Apart from the obvious geographic differences, the studies also differed in the number of reference sites, the number and range of predictor variables and the sampling method and taxonomic resolution (Table 4.1).

Table 4.1. Comparison between model datasets

	Model		
	North Carolina	Skeena Region	Yukon River Basin
Number of reference sites	209	62	61
Taxonomic resolution	Species/Genus	Family	Mainly Genus
Number of taxa	656	70	154
Subsample	> 500	300	200
Range of predictor variables	Basic catchment descriptors, local measurements	Local measurements, GIS catchment descriptors (Table 4.2)	Local measurements, advanced catchment descriptors (Table 4.3)

4.3.2. NC Data

The North Carolina dataset – described in more detail by Van Sickle *et al.* (2005) – consists of 209 reference sites that were sampled between 1983 and 2000. Macroinvertebrates were sampled according to the NCDENR protocol (NCDENR, 2003), which collects a composite sample from 10 habitat targeted samples (2 kick-net samples, 3 bank sweeps, 2 rock or log washes, 1 leaf pack, 1 sand sample, and visual collections from large rocks and logs). Individuals were identified to genus or species level.

Relatively simple predictor variables were used for the NC RIVPACS and ANNA models. Coarse catchment and location descriptors were estimated from electronic 1:25000 topographic maps and included distance from source (DFS), elevation and latitude and longitude. Stream width and depth, as well as percentage of substrate categories (boulder, rubble, gravel, sand, silt) and canopy cover were estimated in the field. We also used 202 independently collected invertebrate samples from the same reference sites to calculate some model metrics.

4.3.3. Skeena Data

Macroinvertebrates within the Skeena region were sampled from riffle habitat with a traveling kick-net method using a 400µm mesh size. Subsamples of at least 300 individuals were identified to family. Seventy invertebrate families were found at 62 reference sites. Reference sites were identified using GIS analysis of stressor gradients in the catchment. Gradients were defined by a Principal Component Analysis (PCA) of GIS layers that quantify potential sources of stress. Mining, agriculture and forest stand age (as a surrogate for logging intensity) were integrated by the catchment area upstream of the site and summarized into stressor gradients analogous to Bailey *et al.* (in press) and Linke *et al.* (in press).

Predictor variables for the reference condition models included on-site measures and GIS variables (Table 4.2). Catchment geology was derived from Geological Survey of Canada data layers. We summarized percent cover of geological strata to three principal axes using PCA.

Table 4.2. GIS and local predictors in the Skeena region for RIVPACS and ANNA. *indicates variables selected in the RIVPACS model $^{+}$indicates variables selected by ANNA

GIS predictors	***Local predictors***
Latitude^{+}	Average Channel Depth (cm) $^{*+}$
Longitude	Maximum Channel depth (cm)
Altitude (m)*	Bankfull width (m) $^{+}$

GIS predictors	*Local predictors*
Slope (m/m)	Channel width (m)
Area in ha	Presence of Coniferous trees
Ecoregion (dummy variable)*+	Presence of Deciduous trees
Total perimeter	Presence of Grasses*+
Mean temperature in January	Presence of Macrophytes*
Mean temperature in June	Presence of Shrubs
Total precipitation in mm	Presence of Pools in the reach
Total snow in mm	Presence of Rapids in the reach+
Percent glaciation in the catchment	Presence of Riffle in the reach
Geology PC1	Presence of Runs in the reach
Geology PC2*	Canopy cover (%)*+
Geology PC3*	

4.3.4. Yukon Data

Sixty-one reference sites in the Yukon Territory (Canada) were sampled in the years 2004 and 2005. The wide geographic area was spread from just north of the Skeena region to 200 km below the Bering Sea. A 400 µm D-net was used to take a 10 m riffle sample. A minimum of 200 invertebrates was identified to the lowest taxonomic level possible, generally genus level. Sites were classified as reference when no disturbance from mining, forestry or roads was present in the upstream catchment.

Table 4.3. GIS and local predictors in the Yukon territory for RIVPACS and ANNA. *indicates variables selected in the RIVPACS model +indicates variables selected by ANNA

GIS predictors	*Local predictors*
Latitude+	Average Channel Depth (cm) *+
Longitude	Average Channel width (m)
Altitude (m)*	Velocity (m/s)
Slope (m/m)	Percentage of bedrock
Area in ha	Percentage of boulder
Total perimeter	Percentage of cobble
Vegetation PC 1	Percentage of gravel
Vegetation PC 2	Percentage of sand
Vegetation PC 3	Percentage of silt
Geology PC 1	Water temperature
Geology PC 2	
Geology PC 3	

Predictor variables for the models included both local observations and GIS variables. Percentages of vegetation type in the catchment (coniferous forest, mixed forest, shrubland, tundra, unvegetated) and bedrock type were summarised using PCA.

4.3.5. RIVPACS model development

For the North Carolina data, the RIVPACS model described by Van Sickle *et al.* (2005) was used. For the other two datasets, we used PCORD (McCune & Mefford, 1999) to cluster reference sites by applying flexible UPGMA to the Bray-Curtis similarity matrix of presence/absence values. Beta was set to -0.1 as recommended by Belbin *et al.* (1993) and Belbin & McDonald (1993). To select model predictors from the pools of environmental variables (Table 4.2, Table 4.3), we ran a stepwise discriminant function analysis in SAS 9 (SAS, 2005).

4.3.6. ANNA model development

ANNA (Assessment by Nearest Neighbour Analysis) accepts the same input data as RIVPACS and delivers similar outputs – expected taxa probabilities that are then used to calculate an observed/expected (O/E) ratio. However, the classification step is circumvented, as ANNA does not operate on groups but calculates a set of nearest neighbours in predicted taxa space for each individual site.

Environmental predictors were selected by running a stepwise principal axis correlation (PCC, Belbin, 1994) on NMDS (non-metric multidimensional scaling, Kruskal, 1964) axes scores. Nearest neighbours for a test site were then selected by ordinating the test site into the reference NMDS space. The number of nearest neighbours in a model was determined by running all models between 6 and 16 nearest neighbours (roughly the range of RIVPACS group sizes) and comparing O/E scores for precision and bias. For detailed information about ANNA model development, see Linke *et al.* (2005).

4.3.7. *Model evaluation*

For both RIVPACS and ANNA, probabilities of occurrence p_{im} of each taxon *i* at each reference site *m* were predicted using the models developed above. Using the taxa with a $p_{im}>0.5$, we calculated the observed/expected ratio OE50. When sites in the reference condition are run through a predictive model, they should be ideally assessed as OE50=1 as one presumes that no taxa loss has occurred. Multiple sites run through the model should result in a mean OE50 of 1 and a narrow standard deviation (SD O/E). This standard deviation has long been used as the currency to assess precision of RIVPACS models (Moss *et al.*, 1999; Hawkins *et al.*, 2000).

We also calculated two measures recently developed to evaluate RIVPACS models (Van Sickle et al., 2005) by establishing best possible and worst possible models (Figure 4.1). Due to the probabilistic nature of predictive models and the limitations of sampling, we would never expect a standard deviation of 0 among reference site O/E values. Instead, the best possible predictive model is dictated by the replicate-sampling variation, which under these circumstances is the only remaining variation. While the true replicate-sampling variation remains unknown in almost all cases, it can be estimated from predicted probabilities of occurrence (Van Sickle et al., 2005):

$$SD_R = \sqrt{\frac{1}{M}\sum_m \left[\frac{\sum_i p_{im}(1-p_{im})}{\left(\sum_i p_{im}\right)^2}\right]}$$

where p_{im} is the probability of occurrence of taxon *i* at site *m* and *M* is the total number of sites. As recommended by Van Sickle *et al.* (2005) we estimated SD_R using p_{im} from the null model.

Good model: close to SD_R

Bad model: not much improvement over SD_{null}

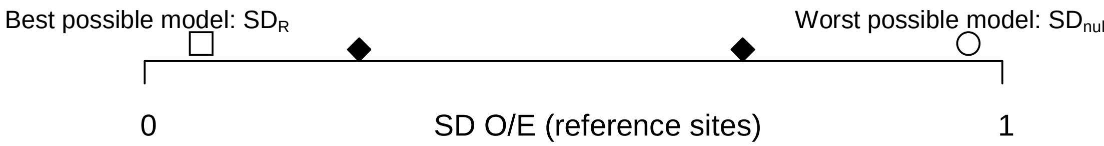


Figure 4.1. Model evaluation using SD_R and SD_{null}.

While SD_R describes a case in which all possible non-sampling related variation is explained by the RIVPACS model, the null model describes the worst possible case: No variation in the data is explained by RIVPACS. The probability of occurrence p_{im} for each taxon under the null model is simply the proportion of all sites at which the taxon is found. The expected value E_{null} - identical at all sites - is calculated by summing all taxa probabilities p_{im} at a site. If all taxa are considered, E_{null} is identical to the mean taxon richness in the dataset. However, if the more common cut-off at p>0.5 is used, only taxa that are present at 50 % of the sites are considered.

To assess the predictive performance of the three RIVPACS and ANNA models, we compared the actual model $SD_{O/E}$ to the SD_R and $SD_{null..}$ Models that show major improvements over SD_{null} and end up close to SD_R are labelled as 'good models'. Models that remain close to SD_{null} are labelled 'weak models' (Figure 4.1).

4.3.8.   Evaluation of taxa prediction

To evaluate how accurately taxa are predicted, we compared two measures. First the total counts of correct predictions across all reference sites were compared. Hereby, a successful prediction is defined as a taxon that has been predicted with a p_{im}>0.5, and is also present in the sample. The second measure of taxonomic accuracy is the number of taxa that were successfully predicted at least once. This delivers an estimator of the variety of taxa to be included in a RIVPACS model: If more taxa correctly predicted at more than one site, the model is potentially sensitive to a greater variety of disturbances.

To test whether the grouping step in RIVPACS has a crucial influence on taxa prediction, we ran an indicator taxa analysis analogous to the one described by Dufrene & Legendre (1997). These indicator values (IVs) measure the strength of association a taxon has with one particular group. For example, if a taxon only occurs in one group at almost every site, its IV is very high. If on the other hand the taxon is equally spread among all the groups, it has no indicator value. After calculating indicator values for the three RIVPACS models it was necessary to estimate the statistical significance of IV's as this varies with the number of occurrences in the dataset. We estimated this significance using 10000 runs of a Monte-Carlo randomisation within PC-ORD (McCune & Mefford, 1999).

We then removed taxa with less than 5 occurrences from the dataset. Using standard RIVPACS group sizes and ANNA prediction clutches, these taxa have no chances of being predicted anyway. The remaining taxa were split in two groups - taxa predicted at least once and taxa that never entered the RIVPACS OE50. We tested for differences between both the IVs and their significance levels using a Kruskal-Wallis nonparametric ANOVA. If the group of taxa that was predicted at least once had a significantly higher median IV and higher significance levels, this indicates that the grouping step has a deciding influence on what is predicted and what is not.

The above definition of a 'correctly predicted taxon' is simplified and not entirely mathematically correct - it ignores the probabilistic nature of RIVPACS and ANNA (models predict the average included taxon at a probability of 0.7-0.8). While the approach outlined above reflects the reality of how the models work – only taxa with $p>0.5$ are included - we also calculated AUC (area under curve) of the ROC (Receiver Operating Characteristic) as a general measure of fit for all the models. If AUC=1, all positive predictions are true positives, indicating a perfect model. The other extreme is AUC=0.5, in which case true and false positives are randomly spread. AUC was calculated for both the NC model building and validation datasets, as well as for the Skeena and Yukon data.

4.3.9. Comparison of O/E(BIODIV)

We calculated O/E(BIODIV) for reference sites in all models as described by Linke & Norris (2003). Taxa predicted at $p<0.5$ are classified as 'rare' or 'unexpected' at the site. O/E (BIODIV) is defined as the ratio between the number of observed 'rare' taxa and the summed probabilities of these taxa. Concordance between RIVPACS and ANNA models strengthens the credibility of O/E(BIODIV).

4.4. Results

4.4.1. Model quality

Model quality for the North Carolina model was good in both RIVPACS and ANNA. Both models were similar and improved considerably over the null model (SD_{null} 0.28). The ANNA model's standard deviation of OE50 was 0.14, while the RIVPACS model was even better at SD_{OE}=0.13. Both models were close to the optimum value of SD_R=0.09 (Figure 4.2). Overfitting was not indicated for either model, as SD_{OE} remained constant in both models when validation samples were run.

Both the RIVPACS and the ANNA model for the Skeena region were very strong. At SD_{OE} of 0.1 and 0.11 respectively, both models were very close to SD_R (0.09). However, in this dataset overall variation between sites was low. Many of the 26 common families occurred at a lot of sites, as testified by the Null model, which has an unusually low SD of 0.14 (Figure 4.2).

The Yukon model was characterised by sites with a low α diversity while having a high β-diversity over the whole dataset (154 taxa at 61 reference sites with an average of 15 taxa per site). This between-sample variability is reflected in a wide SD_R of 0.2 (Figure 4.2). While the

ANNA model (SD_{OE}=0.25) improved over the Null model (SD_{null}=0.3), the RIVPACS (SD_{OE}=0.33) model is even worse than the Null model. This anomaly (usually the Null model is the worst possible model) is caused by the fact that only three taxa are present at p>0.5 across all sites and hence represented in the null model. These taxa were present at almost all sites, which gives the Null model less variation than the RIVPACS model, which predicts more taxa – and occasionally gets them wrong.

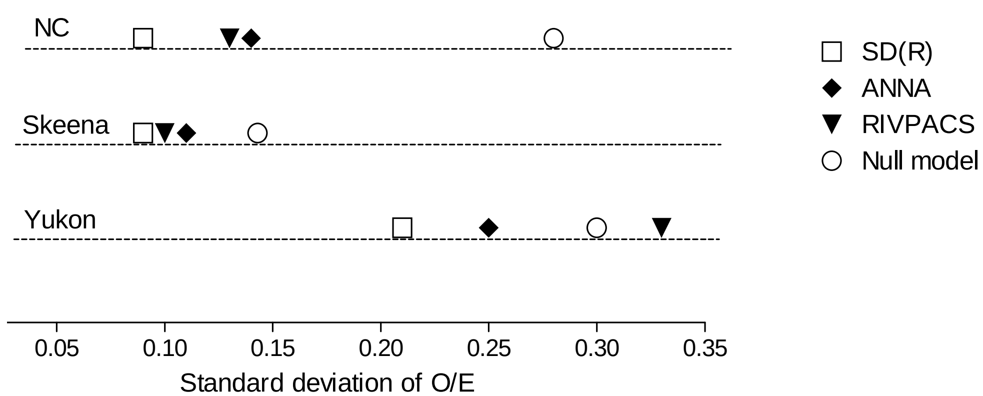


Figure 4.2. Standard deviations for the three RIVPACS and ANNA models, compared with the optimally achievable value SD_R and the Null model. The closer a model's standard deviation is to the optimal value (□) and the further away from the null model (○), the better.

4.4.2. Taxa predictions

For the North Carolina dataset, both RIVPACS and ANNA predicted more than 50% of individual presences at p>0.5. RIVPACS predicted 6754 out of 13600 (49%) total occurrences; ANNA predicted 6830 (50%, Figure 4.3). However, ANNA predicted a total of 231 taxa at least once, 15% more than RIVPACS (196). The validation set produced slightly lower numbers: RIVPACS predicted 6314 occurrences and 24% of the taxa, ANNA predicted 6645 and 27% of the taxa (Figure 4.3, Figure 4.4). The Skeena RIVPACS model predicted a total 730 occurrences of 24 different taxa correctly, 8% less than ANNA (812 out of 1010) (Figure 4.3). ANNA predicted 26 taxa at least once.

As to be expected from the model evaluations, the biggest difference between RIVPACS and ANNA was observed in the Yukon models. While ANNA predicted a total of 377 occurrences at p>0.5 (41% of 912), the RIVPACS model only predicted a total of 206 occurrences correctly (Figure 4.3). The RIVPACS model only predicted 21 taxa (8.3 %) at p>0.5. All of these taxa were predicted at least once by the ANNA model, which predicted an additional 15 taxa at least once.

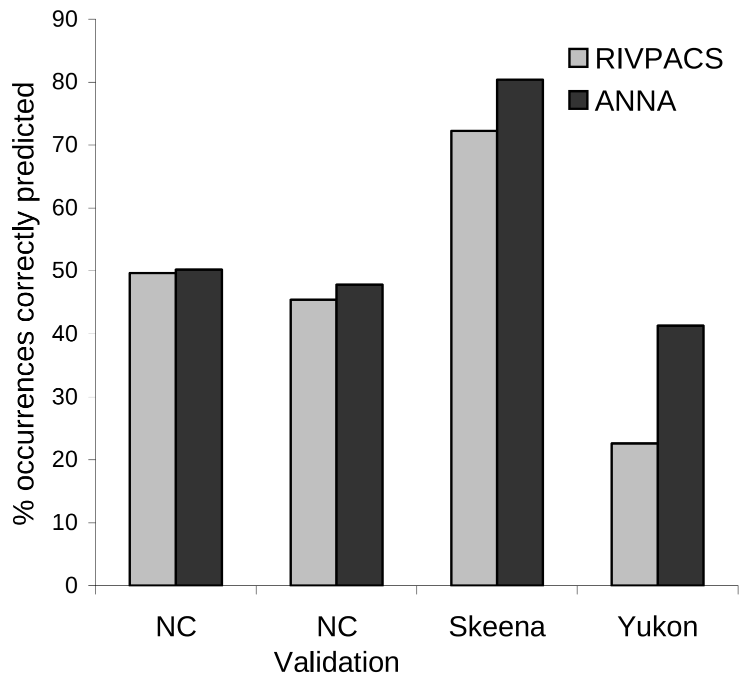


Figure 4.3. Percentage of occurrences correctly predicted using p>0.5 as the cut-off

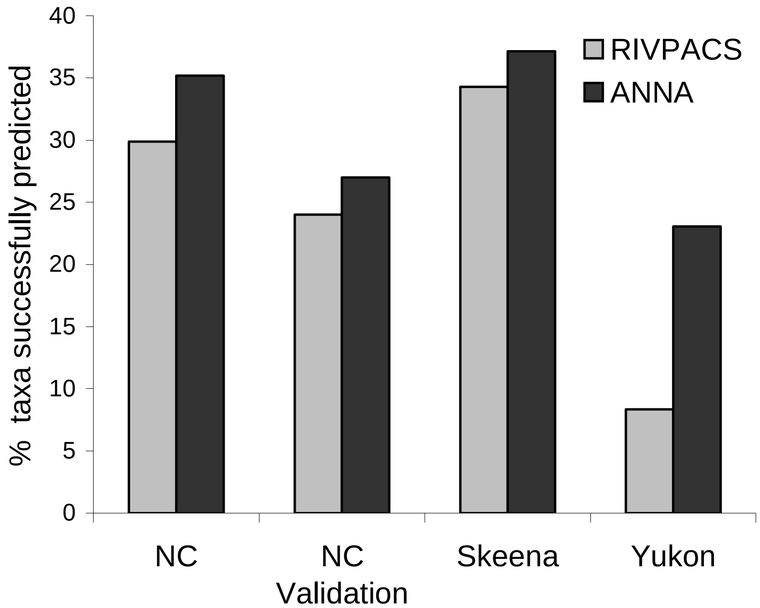


Figure 4.4. Percentages of taxa that were correctly predicted at least once at p>0.5

Although the taxa that were predicted at least once (hereafter referred to as 'the predicted taxa) had higher IVs in the NC dataset, the Kruskal-Wallis test was not statistically significant (χ^2=2.31, df=1, p=0.12). The Monte-Carlo probabilities however were significantly smaller for the predicted taxa (χ^2=18.07, df=1, p<0.0001), indicating that significant indicator taxa are more likely to be predicted than other taxa.

Both the tests for IVs (χ^2=7.474, df=1, p<0.0001) and probabilities (χ^2=12.49, df=1, p<0.0001) were significant in the Skeena dataset. Also in the Yukon dataset, Kruskal-Wallis tests on both the indicator values (χ^2=28.02, df=1, p<0.0001) and the Monte Carlo probabilities (χ^2=23.49, df=1, p<0.0001) were significant. Apart from two very common taxa (present at >75% of sites) and one other exception (*Sperchon* sp., Subclass *Acarina*), only taxa that were significant indicators of a group were ever predicted.

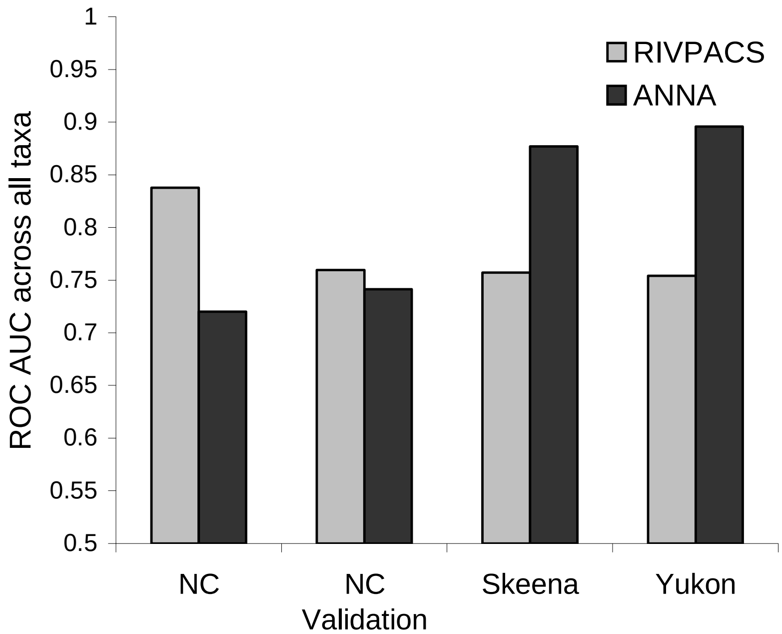


Figure 4.5. ROC AUC (area under curve) for all RIVPACS and ANNA models.

ROC AUC (averaged across all taxa occurrences) showed slight overfitting for the RIVPACS validation dataset: AUC declined from 0.84 to 0.76 (Figure 4.5). AUC of ANNA models remained relatively stable at 0.72 (model building dataset) and 0.74 (validation dataset). For both the Skeena and the Yukon datasets, ANNA had a higher AUC, indicating much better predictions.

4.4.3. Comparing O/E(BIODIV) for both modelling approaches

When comparing O/E(BIODIV) for both modelling approaches, we found a strong correlation in two of the datasets. Both the North Carolina (r^2= 0.69) and the Yukon dataset show high agreement (r^2= 0.61) between the methods (Figure 4.6a,c).

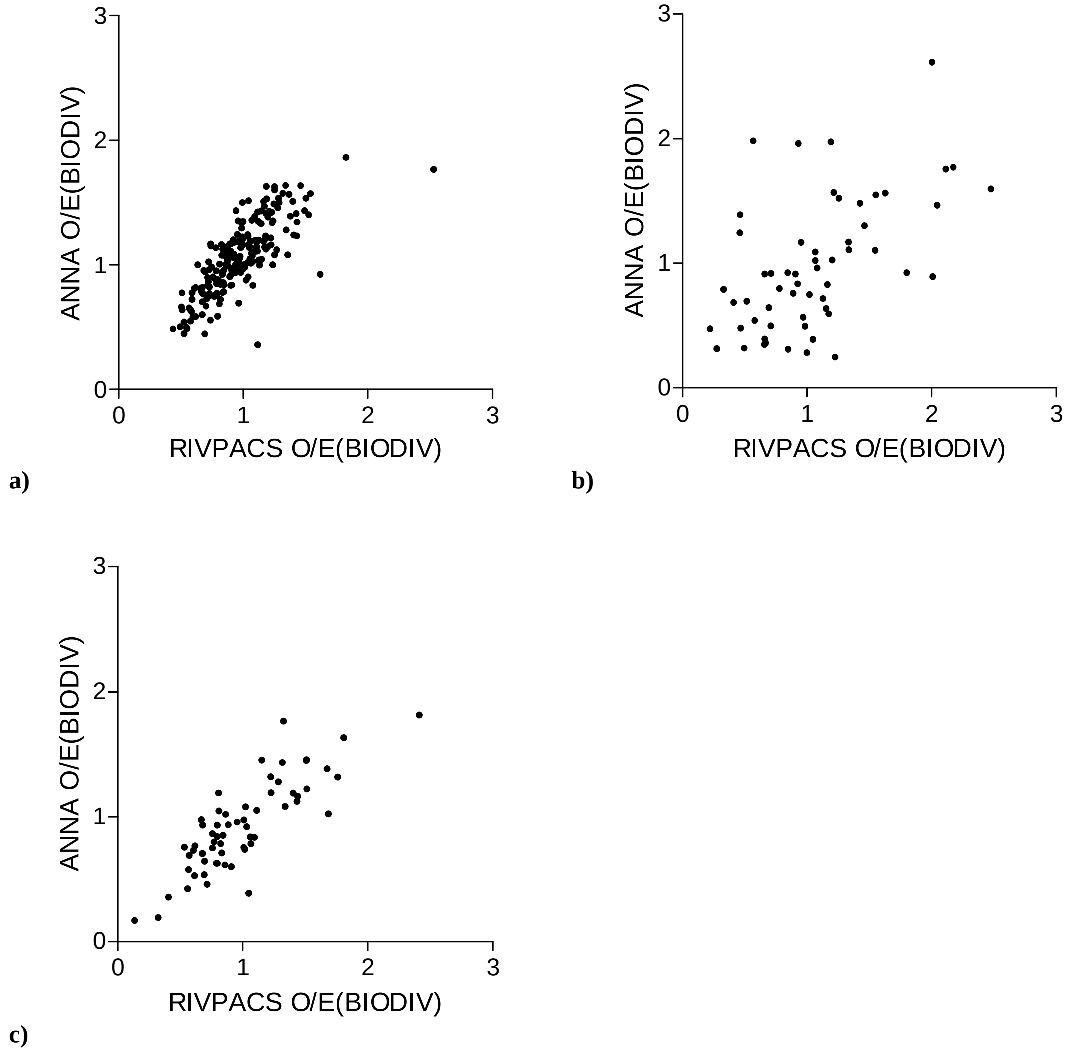


Figure 4.6. Correlations of O/E (BIODIV) between RIVPACS and ANNA for a) North Carolina (r^2= 0.69), b) Skeena (r^2= 0.3) and c) Yukon (r^2= 0.61)

However, a closer look at the data revealed that while both observed and expected values of O/E (BIODIV) for the North Carolina dataset was very similar between the models, the Yukon O/E (BIODIV) was completely driven by the collected taxa in both models. The Skeena O/E (BIODIV) showed considerable variation between RIVPACS and ANNA model, with only 30% of the variation explained.

4.5. Discussion

This study highlights the need for a taxon-specific evaluation of predictive models in both bioassessment and biodiversity assessment. For two of the three datasets, RIVPACS and ANNA models showed very similar model properties when evaluated in the conventional way (SD_{OE}), but differed in both the number of occurrences correctly predicted and the number of taxa that were entered in the OE_{50} –thus lowering sensitivity for RIVPACS and ANNA style bioassessment. It also lowers confidence in a site-specific rarity measure.

4.5.1. Model validation and ROC area under curve (AUC)

ROC AUC showed slight overfitting for RIVPACS models, yet not for ANNA models (Figure 4.5). However, overfitting did not reflect on the key measures discussed below. Model quality (as measured by SD O/E) did not differ between the NC calibration and validation sets. Also, despite slightly lowering the predictive success at the taxon level (Figure 4.3, Figure 4.4) in the validation, the overall patterns in the NC dataset remained identical between both datasets. We can therefore conclude that overfitting is not a critical issue for this study.

AUC is similar between RIVPACS and ANNA for the NC validation dataset, but is considerably higher for both the ANNA models in the Skeena region and the Yukon. While the higher values indicate an overall better model fit, they are not directly relevant for RIVPACS and ANNA assessments because the models only consider taxa with $p>0.5$.

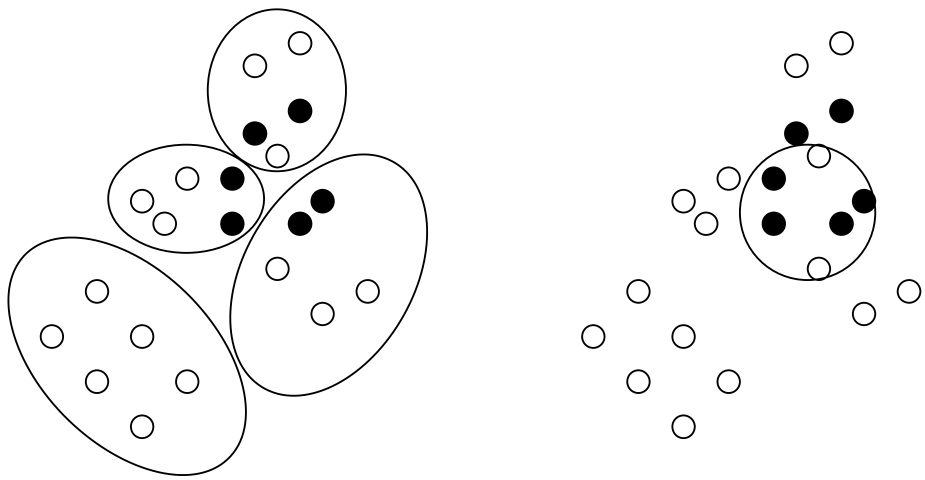
4.5.2. Implications for bioassessment

Differences in the number of successfully predicted taxa (Figure 4.4) - in conjunction with the results of the Kruskal-Wallis tests - clearly demonstrate the influence of the grouping step in RIVPACS on the prediction of taxa. While SD(O/E) is almost identical for the North Carolina RIVPACS and ANNA models, 35 taxa were never predicted using RIVPACS. The ANNA model for Skeena only predicted two additional taxa compared to the RIVPACS model, but the Yukon models showed the biggest discrepancy.

Because taxa that are not predicted but are present at p>0.5 drop out of the OE50 score used in Australia and the USA (Hawkins et al., 2000), this results in a potential loss of sensitivity. In the Yukon RIVPACS model for example, only 22 genera would cause a drop in O/E if they were predicted but missing in the sample. If human induced stress caused a local extinction of one of the remaining 132 taxa, it would not be picked up by the RIVPACS model – the O/E would stay at close to one and the site would remain in a higher band. Thus, a model that can predict more taxa correctly is more sensitive.

ANNA consistently predicted more taxa correctly across the datasets and also succeeded in predicting more single presences of taxa at sites compared to the corresponding RIVPACS models. The Kruskal-Wallis tests on the indicator values demonstrate that indicator taxa are significantly more likely to be predicted by a RIVPACS model. This makes intuitive sense: In order to be predicted at all, a taxon has to be present at 50% of sites at a reference group. An extreme example is depicted in Figure 4.7. Despite being present at very similar sites, the example taxon will never be predicted as its highest single group probability is 40%. In ANNA, however, the taxon can be predicted if its distribution follows a gradient described by the predictor variables (Figure 4.7b).

In our data, this effect is demonstrated most clearly in the Yukon dataset, where most of the 22 taxa predicted by RIVPACS were indicator taxa, while the 14 additional taxa that were predicted by ANNA had lower IVs. This finding is especially troubling in the light of the studies by Moss *et al.* (1999) and Podani (1997), who found that variations in the nature of the distance measures and clustering methods can have profound effects on clustering. Also, trials for this study showed that adding two random taxa to 10 sites in the Yukon dataset resulted in a major shift of group structure and high IVs for the made-up taxa. (Linke, unpublished data)



a) b)

Figure 4.7. Example of a taxon that is not predictable by RIVPACS (a) but predictable using ANNA (b). Dots (●) indicate sites where the taxon is present. Circles (○) indicate sites where the taxon is absent. Distance between sites reflects environmental similarity.

Despite ANNA performing a lot better on the Yukon dataset than RIVPACS in terms of both traditional evaluation methods (SD_{OE}) and taxa prediction, it is questionable whether any of the two Yukon models would be acceptable for a bioassessment program. Over 100 taxa (~ 65% of all taxa) have five or less observations in the dataset. Whether these taxa occur at few sites or they are just not detected at the remaining sites due to the low local abundances – it will not be possible to predict them. In cases like the Yukon dataset where α-diversity at the site is extremely low compared to β-diversity across the dataset, approaches that do not predict probabilities of occurrence of single taxa might work favourably to RIVPACS or ANNA. Regression approaches like the ones by Bailey *et al.* (1998) and Linke *et al.* (1999), metric nearest neighbour methods like Prins & Smith (2005) or IBI stratified by ecoregion (Barbour et al., 1996) are possible alternatives.

4.5.3. Implications for biodiversity assessment

Despite the intuitive appeal of an index of site-specific rarity as discussed by Linke & Norris (2003), this study shows that the concept can only work if a maximum of the variation in the data is explained by the underlying model. In the case of the North Carolina data or similar datasets, OE(BIODIV) could potentially be applied: model quality is close to the maximum achievable value and despite the obvious influence of the grouping step in RIVPACS (Figure 4.4), both models deliver the same results in terms of O/E (BIODIV). The main difference between the models is that the number of 'unexpected' taxa is slightly higher in the RIVPACS model – ANNA predicts more common taxa. However, the ratio between observed and expected rare taxa OE(BIODIV) is robust despite the slightly different predictions.

For both the Skeena and the Yukon models, we have lower confidence in the applicability of O/E(BIODIV) relative to the NC models. In the Skeena models, we observed a strong disagreement between the two OE(BIODIV) - despite both approaches performing extremely well in the internal validation. Skeena RIVPACS and ANNA models declare different taxa as rare, which in this case leads leading to different values of O/E(BIODIV).

In the Yukon model, both RIVPACS and ANNA OE(BIODIV) show a similar agreement to the NC models. Yet a closer look at the data reveals that in these models the aim of OE(BIODIV) – a site specific rarity index – has not been achieved. With most genera never predicted at $p>0.5$, these taxa are classified as rare and therefore count towards OE(BIODIV). As only 22% occurrences in 13% of genera were predicted, the remaining observations count towards O/E(BIODIV). In this case, the site specific rarity index is transformed into a plain measure of taxa richness – which demonstrates a second shortcoming of OE(BIODIV).

Overall, ANNA seems more suitable for calculating O/E(BIODIV). The Kruskal-Wallis tests, as well as the numbers of correctly predicted taxa (Figure 4.3) clearly show that the grouping step in RIVPACS limits which taxa are predicted. The reason is that the structure of the groups and therefore the sites that have an influence on the prediction is formed by the taxa alone. The nearest neighbours in ANNA are chosen based on environmental similarity. Therefore, ANNA is limited by the explanatory power of the environmental gradients and the quality of the model building process, whereas in RIVPACS an *a priori* decision about what taxa will be best predicted is cast after the grouping step (Figure 4.7). Obviously, the grouping step influences the kind and the number of the taxa classified as rare and considered in O/E(BIODIV) - only avoided by classifications with strong gradients and a large sample size as shown by the NC data.

4.5.4. Conclusions

While the RIVPACS grouping step might not affect the precision of bioassessment applications, it limits the taxa that are predictable. When calculating the standard metric (OE50), taxa that are not indicators for the classification groups often drop out of the assessment, therefore potentially reducing sensitivity. The percentage of predicted taxa and the percentage of predicted occurrences should therefore be used as model criteria in further studies.

Under-prediction of taxa is even more consequential for calculating O/E(BIODIV). Whereas taxa below $p<0.5$ drop out of OE50, they enter O/E(BIODIV). Thus, a taxon that is not predicted because of RIVPACS or ANNA modelling error changes the assessment by increasing O/E(BIODIV). For this reason, O/E(BIODIV) can only be applied meaningfully if a large amount of variation is explained by the model. As it is generally difficult - if not impossible - to assess how much of the explained variation is enough, an objective output of O/E (BIODIV) will be hard to achieve.

Chapter 5. Irreplaceability of river networks: Towards catchment based conservation planning

This chapter is ready for submission to the journal ‘Journal of Applied Ecology’

S. Linke, R.H. Norris and R.L. Pressey. Irreplaceability of river networks: Towards catchment based conservation planning

5.1. Abstract

In modern conservation planning science, irreplaceability measures the conservation value of a planning unit by assessing the importance of the unit to fulfil conservation targets. We adapted a complementarity-based method to estimate irreplaceability for river systems. After dividing the Australian state of Victoria into 1854 subcatchments based on a 3 arcsecond digital elevation model, we successfully modeled distributions of 400 benthic macroinvertebrate taxa using generalised additive models. We calculated the minimum area required to protect all taxa using three different selection metrics to determine the most efficient heuristic complementarity algorithm. These metrics were modified to consider the entire upstream catchment. A summed rarity algorithm, corrected for upstream area, proved to be the most successful metric, requiring 100 000 hectares less than the second most efficient algorithm. We calculated irreplaceability by running the algorithm 1000 times and randomly removing 90% of the catchments each run. From these runs, we estimated two metrics: *f* (the frequency of selection across 1000 runs) and *average c* (contribution to conservation targets). Four groups of catchments were identified: a) catchments that have high contributions and are always selected; b) catchments that have high contributions and are not always selected; c) catchments that are always chosen but do not contribute many taxa; d) catchments that are rarely chosen and do not contribute many taxa. *Summed c*, the sum of contributions over 1000 runs, was chosen as the indicator of irreplaceability, integrating the frequency of selection and the number of taxa protected. This indicator showed a bias for upland and lower-order coastal catchments, which is an effect of the upstream-protection rule in the algorithm.

5.2. Introduction

While freshwater biodiversity is being lost rapidly all over the globe (Moyle & R.M., 1994; Abell, 2002; Barmuta, 2003), calls for establishing freshwater protected areas are increasing (Saunders *et al.*, 2002; Kingsford *et al.*, 2005). Matters have slightly improved since Cullen & Lake (1995) raised concern that the biodiversity movement is paying little attention to freshwater systems. However, river conservation science is still lagging in quality and quantity compared to terrestrial assessments of conservation value (Daniels *et al.*, 1991; Freitag *et al.*, 1997; Root *et al.*, 2003) and terrestrial applications of systematic conservation planning (Margules & Pressey, 2000; Abell, 2002; Sarkar *et al.*, 2002; Cowling *et al.*, 2003a; Pierce *et al.*, 2005), which have been steadily building since Kirkpatrick's first complementarity algorithm (Kirkpatrick, 1983).

Only a handful of rivers that have reserve status was specifically designed as freshwater reserves. These include the Nahanni National reserve in Canada (Saunders et al., 2002) and the Pacaya-Samiria National reserve in Peru (Bayley et al., 1991). Most of these are protected in the framework of a terrestrial reserve system. This has been labeled as insufficient by a number of authors (Lake, 1980; Maitland, 1985; Skelton *et al.*, 1995). One obvious limitation of this approach is that terrestrial conservation planning usually does not consider aquatic taxa as targets, which are thus under-represented (Nilsson & Gotmark, 1992). Also, specific threats to freshwater ecosystems and their high levels of connectedness are not necessarily considered in the selection of terrestrial reserves (Angermeier & Winston, 1999; Filipe *et al.*, 2004).

The most fundamental difference between terrestrial and riverine conservation planning is the required spatial configuration of potential protected areas. River networks are connected systems, both laterally and longitudinally. While terrestrial conservation planning is increasingly addressing issues of connectivity when dealing with metapopulations (Cabeza, 2003; Fischer & Church, 2003), the nature and scale of connectivity are different in freshwater systems. Sections of a river can be affected by activities hundreds or even thousands of kilometres upstream. Therefore, in accordance with Hynes' (1975) paradigm that 'the valley rules the stream', upstream areas must be considered when estimating conservation value or developing a system of protected areas (Moyle & Sato, 1991; Puth & Wilson, 2001; Crivelli, 2002; Collares-Pereira & Cowx, 2004). Some authors acknowledge that exceptions have to be made for very large streams because whole catchment protection is too difficult to achieve (Saunders *et al.*, 2002; Collares-Pereira & Cowx, 2004). While recent studies also note that cross-catchment and downstream influences should be considered (Pringle, 2001; Yates & Bailey, in press-a), we propose here - for simplicity's sake - to only incorporate upstream protection into a potential measure of conservation value.

Many freshwater conservation efforts are based around single taxa, including the whole-catchment Pacaya-Samiria National reserve in Peru (Bayley et al., 1991), which was designed to protect the world's largest freshwater fish *Arapaima gigas* (Schinz 1822). However, assemblage-based conservation strategies are more ecologically sound and cost-effective (Franklin, 1993; Angermeier & Winston, 1997). Most assemblage-based assessments of aquatic conservation value rely on indices. Filipe *et al.* (2004), for example, assigned a conservation value to river reaches by multiplying the probability of occurrence of each taxon with its rarity. Angermeier & Winston (1997) developed an Index of Centers of Density which helps to identify regionally rare taxa and assess the relevance of sites as sources for potential recolonisation. While the British SERCON (System for Evaluating Rivers for Conservation, Boon, 2000; Boon *et al.*, 2002) integrates richness, rarity and naturalness into a score, other authors use only richness as a measure of conservation value (Williams et al., 2004). We will not be using an index, but introduce a method based on complementarity, the key principle underlying all systematic planning methods in the terrestrial and marine realms (Margules & Pressey, 2000; Sarkar *et al.*, in press). More specifically, we adapt complementarity to recognize upstream connectivity of freshwater systems.

After widespread use of indices for measuring conservation value in the 1970s (see Justus & Sarkar (2002) for a comprehensive review), complementarity was a key advance in the 1980s (Pressey, 2002). Complementarity-based selection algorithms require quantitative conservation targets for biodiversity surrogates such as species or habitat types. The role of these methods is to minimize the total cost of achieving all targets or to maximize the number of targets achieved within limits of total cost (Pressey *et al.*, 1997; Faith & Walker, 2002). In doing so, they recognise the contributions of any established reserves and identify sets of additional areas that are highly complementary in the features they contain, relative to the targets. Scoring indices are highly inefficient at achieving the same objectives, specifically because they do not recognise complementarity between areas (Kirkpatrick, 1983). Complementarity-based reserve design projects have been conducted on all continents (Williams *et al.*, 1996; Pressey & Taffs, 2001; Sarakinos *et al.*, 2001; Cowling *et al.*, 2003a; Faith *et al.*, 2004) mainly for terrestrial conservation but increasingly to design marine reserves (Stewart & Possingham, 2005).

Many complementarity algorithms have addressed a minimum set problem (Margules et al., 1988), trying to determine the smallest number of sites that would cover all surrogates. Near-minimum sets have been calculated by stepwise heuristic algorithms (Csuti *et al.*, 1997; Pressey *et al.*, 1997) and simulated annealing (Possingham et al., 2000). Some studies have also used optimising algorithms from operations research and location science to find true minimum sets (Camm et al., 1996). However, near-minimum or true minimum (hereafter "minimum") sets represent only single solutions to the problem of achieving targets for all features. To explore the options for achieving targets, Pressey *et al.* (1993; 1994) and Ferrier *et al.* (2000) proposed a new measure - ‘irreplaceability’. The operational definition of irreplaceability has two aspects (Pressey et al., 1993): 1. the likelihood that an area will be required as part of a conservation system that achieves all targets; and 2. the extent to which the options for achieving all targets are reduced if the area is unavailable for conservation.

Because measurement of irreplaceability is a combinatorial problem, it cannot be calculated exactly for large regional datasets and has to be estimated (Jacobi et al., in press). Methods for estimation can be classified into three groups: 1. stepwise heuristic estimators (Pressey et al., 1994); 2. statistical estimators (used in C-Plan; Ferrier et al., 2000); and 3. methods that explore multiple possible site combinations (Rebelo & Siegfried, 1992; Possingham *et al.*, 2000; Tsuji & Tsubaki, 2004). While at first glance the problem seems trivial, estimating irreplaceability can be deceptively difficult (Jacobi et al., in press). Only some areas will default to irreplaceabilities of one because they have unique features. Areas can have high irreplaceability values for other reasons, depending on the way targets are defined and features distributed across regions. More generally, it is important to measure irreplaceability of all areas across the range of values from one to zero. Here we present a new estimation method using a bootstrapped heuristic algorithm that allows us to incorporate upstream connectivity of catchments.

One of the few systematic conservation planning approaches for rivers to date (Roux et al., 2002) used landscape attributes as biodiversity surrogates. Other studies propose river classes as features to be represented (Fitzsimons & Robertson, 2005) – similar to the land classes used in terrestrial assessments by Lombard *et al.* (2003) and Pressey *et al.* (2000)(2000). Although we do not believe that perfect taxonomic data are necessary for conservation planning, we decided to set targets based on actual taxa, as the use of environmental surrogates has been heatedly debated (Araujo *et al.*, 2001; Brooks *et al.*, 2004a; Brooks *et al.*, 2004b; Cowling *et al.*, 2004; Higgins *et al.*, 2004; Molnar *et al.*, 2004; Pressey, 2004). To avoid negative spatial bias in unsampled areas (Wilson et al., 2005b), we modeled distributions of our target organisms analogous to Clark & Slusher (2000) and Richardson & Funk (1999)

The main aim of this study is to develop a method to estimate the conservation value of river systems subject to the following requirements:

1. The method is based on complementarity and identifies minimum sets and estimates irreplaceability values;
2. Catchment effects are recognised and accounted for by incorporating whole-catchment protection into the reserve selection algorithm;
3. Taxa are the features to be targeted and represented. When taxonomic records are not available, probabilities of occurrence are estimated.

We demonstrate our approach using data on benthic riverine macroinvertebrates in the State of Victoria, in southern Australia.

5.3. Methods

5.3.1. Study area

Victoria is about 227 600 km^2 and covers a wide variety of landforms and climatic conditions. The western plains is a semi-arid region, temperate rainforest is the main natural land cover in coastal areas, and the Victorian part of the Snowy Mountains is an alpine region above 2000 m of altitude. This variation is ideal to develop and test new conservation methods. The northwest corner of Victoria was omitted for two reasons. First, parts of it no have water at all. Second, the main stem of the River Murray runs through part of this corner. It has a distance to source of over 1000 km in this corner of Victoria with headwaters also in other states. We did not have access to comparable data for these upstream areas. Also, management across state boundaries requires resolution of issues past the scope of this publication. We therefore excluded the Victorian northwest and interstate regions from the analysis. We included parts of the Murray-Darling basin that have their headwaters in Victoria, as well as a small part of the south-eastern corner of neighbouring New South Wales (Figure 5.4).

5.3.2. Planning units

While studies in terrestrial conservation planning often use equal sized grid cells for their analysis (Gaston & David, 1994; Freitag & Van Jaarsveld, 1998; Araujo, 1999; Sarkar *et al.*, 2002; Cowling *et al.*, 2003a; Warman *et al.*, 2004a), we accounted for the connected nature of rivers and the natural boundaries of influence by delineating subcatchments as planning units. We used the SRTM 3 arcsecond DEM (van Zyl, 2001) and patched holes smaller than 3 pixels using 3-DEM (Horne, 2006). A total of 1854 Subcatchments was then delineated using ArcHydro (Maidment, 2002) within ArcGIS 9 (ESRI, 2002). These entities are hereafter referred to as ‘subcatchments’, in contrast to ‘catchments’, which describes the entire area upstream of a river link. If not specified otherwise, all integration of raster layers to catchments was performed using Hawth’s Tools (Beyer, 2004).

5.3.3. Benthic macroinvertebrate data and environmental predictors

Benthic macroinvertebrates were used as targeted features to calculate minimum sets and irreplaceability of subcatchments. Invertebrates were collected using a D-net kick sample (250 μm mesh) and sorted in the field (see Marchant & Hehir, 2002; Metzeling *et al.*, 2003) until at least 200 individual specimen were recovered. Invertebrates in 222 subcatchments were identified to species where possible, amounting to a total of 1065 taxa.

Predictor variables for modelling benthic invertebrates in unsampled catchments can be categorised into four major groups (Table 5.1).

1. Location of the subcatchment in the landscape. These included latitude and longitude and distance from the outflow sea. Total catchment area upstream was calculated using ArcHydro (Maidment, 2002). We also calculated mean elevation and range in elevation of the whole catchment upstream. More locally, we calculated the mean elevation of subcatchment, as well as the standard deviation of elevation as a measure of topography.

2. Climate. These were taken from the ANUCLIM model (Houlder et al., 2000). Average rainfall was calculated for the subcatchment, but rainfall was also integrated across the whole catchment area upstream. Temperature from ANUCLIM was averaged at the subcatchment level.

3. Landform. Slope was derived from the SRTM DEM using Spatial Analyst in ArcGIS 9 (ESRI, 2002). To approximate slope from the DEM in geocoordinates, the slope was approximated using a modified z-factor:

   z-factor = 1/ (113200 * cos(*Latitude in radians*)) (ESRI Support Center, 2005). Mean and standard deviation slope were summarised both at the catchment and the subcatchment level. More complex landform categories were calculated using the 'toposhape' module in IDRISI Kilimandjaro (Clark Labs, 2004). These included ridge, ravine, concave slope, convex slope, and saddle. Percentages were summarized by subcatchment.

4. Geology and vegetation. Vegetation growth category and vegetation density were calculated using John Carnahan's spatial vegetation bibliography (AUSLIG, 1991). After converting the coverage to a 3 arc-second raster layer, we recoded the dominant

growth classes to 0 (grasses), 1 (low shrubs), 2 (tall shrubs/low trees) and 3 (medium/tall trees). Vegetation density classes were recoded to 0 (<10% density), 1 (10-30%), 2 (30-70%) and 3 (>70%). The new values were averaged by subcatchment. Percentages of sandstone, siltstone, limestone, acid volcanic, and basic volcanic soils were derived from the digital version of the 1:2.5 million Geology of Australia Map (Palfreyman *et al.*, 1976; Bureau of Rural Sciences, 1991) by using the Intersect tool in ARCToolbox (ESRI, 2002).

Table 5.1. List of predictor variables for modelling invertebrate taxa

Location in landscape	Climate	Landform	Geology and vegetation
Latitude	Local average rainfall in the subcatchment	Slope in the catchment (mean, standard deviation)	Vegetation growth category
Longitude	Total rainfall in the catchment	Slope in the subcatchment (mean, standard deviation)	Vegetation density
Total catchment area	Average temperature in the subcatchment	Percentage of each landform in the subcatchment: • Ridge • Ravine • Concave slope • Convex slope • Saddle	Percentage of each geological type in the subcatchment: • Sandstone • Siltstone • Limestone • Acid volcanic • Basic volcanic
Elevation in the catchment (mean, range)			
Elevation in the subcatchment (mean, standard deviation)			
Distance from sea			

5.3.4. Using generalised additive models to model distributions of taxa

Using the predictors in Table 5.1, we derived a predictive relationship for each taxon using Generalized Additive Models (GAMs, Yee & Mitchell, 1991; Hastie & Tibshirani, 1999). Unlike linear or logistic regressions, which assume a monotonic response (Hastie &

Tibshirani, 1999), GAMs can fit curvilinear relationships, such as optimum curves (Austin, 2002; Yuan, 2004a). This ability to model both monotonic and Gaussian response curves (Bio *et al.*, 1998; Ejrnaes, 2000) makes GAMs appropriate statistical models for ecologists (Austin, 2002).

When building the GAM models, we decided not to withhold a validation dataset. The main reason was that the rarer taxa would have been impossible to validate based on only one or two observations. Considering that we wanted to estimate conservation values of subcatchments instead of presence/absence of single taxa we were comfortable with this decision. However, we took two measures to avoid overfitting. First, we removed taxa with less than ten observations in the entire dataset. Second, analogous to the modelling approach by Yuan (2004a) we ran a principal components analysis (PCA, Pearson, 1901) to eliminate correlated variables and reduce the number of predictors. The predictors that showed the highest correlation with the first six principal axes were used to build a stepwise GAM for every taxon. The criterion used in the stepwise selection was AIC (Akaike's information criterion), which works analogously to adjusted r^2 by penalising for added variables. To reduce overfitting, the maximum number of variables was set to two and the spline interpolation was limited to 2 degrees of freedom, effectively allowing only one optimum.

Our measure of modelling success was AUC (area under curve) of the ROC (Receiver Operating Characteristic). AUC is a measure of the ratio of true positives to false positives. While an AUC of 1 shows perfect prediction, 0.5 indicates a random spread of true and false positives. Based on the classification by Boyd *et al.* (2005, Table 5.3), we set the cut-off for successful predictions to 0.6. We predicted probabilities of occurrence of 400 successfully modelled taxa for all 1854 subcatchments. Probabilities were then converted to presence/absence at a threshold of 0.5.

5.3.5. Minimum sets and irreplaceability

Our representation goal was to design a network capturing all 400 successfully modelled taxa at least once. The reserve selection problem is therefore analogous to an SSCP (species set covering problem, Possingham *et al.*, 2000; Williams *et al.*, 2005; Jacobi *et al.*, in press). The parameter to optimise (minimise) is the number or total area of subcatchments in which all taxa are represented. We used three heuristic approaches to estimate a minimum set. Heuristic approaches were chosen because they consist of simple rules which makes them easy to follow (Pressey *et al.*, 1999; Margules & Pressey, 2000; Cabeza & Moilanen, 2003) and are easily modified to include rules for catchment connectivity.

All three algorithms were modified to account for the connected nature of rivers. In a lotic setting, it is clearly not enough to protect isolated subcatchments. To ensure that upstream disturbances would not affect potential aquatic reserves, we introduced a rule that the entire catchment upstream of any selected subcatchment had to be protected. In the reserve design algorithm, single subcatchments were forbidden configurations if they were not headwater catchments (Figure 5.1). Selection of a non-headwater subcatchment for conservation also selected the subcatchments upstream (Figure 5.1c). All taxa found upstream at a single subcatchment were determined using a propagation algorithm based on the ARC Hydro network (Maidment, 2002). This whole-catchment taxa list was the input for the three heuristic algorithms. Additionally because catchment sizes varies by three orders of magnitude, headwater catchments to the mouth of the Snowy River- the measures of number and rarity used in the selection process have to be corrected for the total area protected.

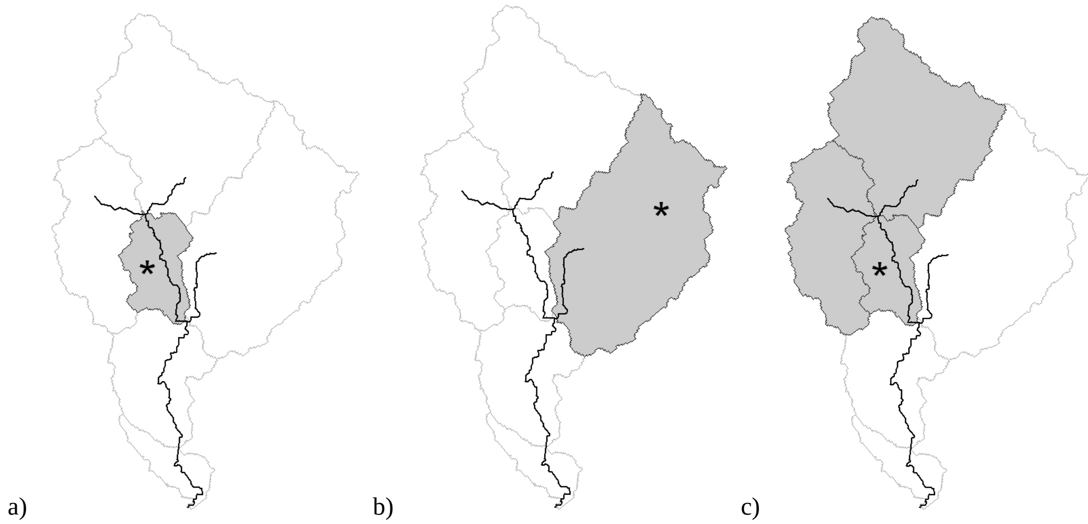


Figure 5.1. Forbidden and allowed configurations in the complementarity algorithm. Protected subcatchments are highlighted by shading. Isolated subcatchments that are not headwater catchments are forbidden configurations (a). Single headwater subcatchments are allowed configurations (b) as are whole catchments that include the subcatchment of interest (c), in this case marked with an asterisk

The richness-based greedy heuristic algorithm (Kirkpatrick, 1983; Pressey, 2002) starts by finding the catchment with the highest number of taxa. The catchment and the associated taxa are then removed from the dataset and richness is re-calculated, so successive selections will find catchments with the highest number of previously unrepresented taxa. The procedure continues until all taxa are in at least one selected catchment. We corrected the richness measure by dividing it by the size of the catchment to incorporate cost. The coefficient for selecting catchments is:

$$c = n / area \qquad \text{(Equation 1)}$$

where c is contribution to targets, n is number of taxa in the catchment, and *area* is hectares covered by the catchment.

The different catchment sizes meant that the algorithm did not have to choose between catchments with equal values of c, making tie-breaking rules (Pressey *et al.* 1997) redundant. This algorithm is the first and most simple heuristic solution to the reserve design problem.

However, it is also the least efficient (Underhill, 1994; Csuti *et al.*, 1997; Pressey *et al.*, 1997).

Based on the findings of Csuti *et al.* (1997), we used a progressive rarity algorithm (Margules et al., 1988) as the second heuristic. In this method, also used in ResNet (Garson *et al.*, 2002a; Sarkar *et al.*, 2002), the first catchment selected is identified by the rarest taxon, with rarity here divided by the area of the catchment (Equation 2).

$$c = \frac{1}{f} / area \qquad \text{(Equation 2)}$$

where c is the selection metric, f is the frequency of the taxon (recalculated after every iteration) in the entire dataset, and *area* is hectares covered by the catchment. The selected catchment and taxa are removed and the catchment with the next largest c selected. Again, the process is repeated until all taxa are in at least one selected catchment. No tie-breaking rules were needed.

The third algorithm is based on total summed rarity, adjusted for area (Equation 3), analogous to the algorithms used by Rebelo & Siegfried (1990; 1992).

$$c = \sum \frac{1}{f} / \text{area} \qquad \text{(Equation 3)}$$

where c is contribution to targets, summed across all taxa in the catchment and corrected for area, f is frequency of the taxon in the entire dataset, and *area* is hectares covered by the catchment. As before, c is recalculated as selections proceed until all taxa are in at least one selected catchment, and no tie-breaking rules were needed.

To calculate irreplaceability while still incorporating catchment area, we developed a new approach that is not based on a statistical estimator (Ferrier et al., 2000), but is closer to Rebelo & Siegfried's approach (1992) of re-running a heuristic selection process from different starting points. Instead of choosing different seed catchments, we removed a fixed percentage of catchments each time we ran the algorithm. This is a simulated analogy to the real-world scenario that drove the development of irreplaceability measures: a certain percentage of the area is made unavailable. Using the summed rarity algorithm (which was more efficient than the others), we removed catchments randomly 1000 times and ran the selection algorithm each time. We repeated this process for three levels of removal: 50%, 70% and 90%. For each level of removal, we then derived two measures from the 1000 minimum sets of catchments. The first was the *f* – the frequency of selection, varying from zero to 100% of the minimum sets. The second was the *average c* (see Equation 3) or the average contribution to targets of each catchment over the 1000 runs.

We believe both properties are important to planning scenarios – for example catchments with a small *f* are still needed for full representation if they contain endemic taxa. When looking for biodiversity hotspots, catchments with a high *average* c will achieve highest returns. To combine both properties into a single index of irreplaceability, we use *summed c* – the sum of *c* over all runs a catchment was selected in.

$$\textit{summed } c = \sum_{i=1}^{n} c$$

where *n* is the number of minimum sets with the catchment present and *c* is the contribution that the catchment has in the *i*th minimum set. This estimator will hereafter be termed 'irreplaceability'.

5.4. Results

The first six principal components of the predictor variables PCA explained 71 percent of the total variation in the dataset. We selected the six variables with the highest loadings as predictors for the generalised additive models. These were:

- Average slope in the subcatchment
- Catchment area upstream
- Percent of ravines in the catchment
- Vegetation density
- Percent of limestone in the catchment
- Percent of sandstone in the catchment

Correlated variables that had high loadings (>30%) on the six main axes are summarised in Table 5.2.

Table 5.2. Predictors with high loadings on the principal axes. *indicates that predictors were used for modeling

PC1	PC2	PC3	PC4	PC5	PC6
Average slope in the subcatchment*	Catchment area upstream*	Percent ravines*	Vegetation density category*	Percent limestone*	Percent sandstone*
Standard deviation of slope in the subcatchmemt	Total rainfall in the catchment	Percent ridges	Percent limestone	Local temperature	Percent acid volcanic
Average elevation in the subcatchment	Range in catchment elevation	Catchment area upstream	Percent convex hills		
	Latitude				
	Percent sandstone				

Out of the 452 macroinvertebrate taxa with more than 10 occurrences, 400 taxa were successfully modelled. The other 52 taxa remained below the ROC AUC of 0.6 or did not produce any predictions above 50 percent probability of occurrence. Out of the 400 successfully modelled taxa, most predictions were good to very good, with only ten percent below 0.7 (Table 5.3).

Table 5.3. Prediction success for the modelled taxa

Prediction category (Range of AUC in percent)	Percentage of taxa in category
Average (0.6-0.7)	10
Good (0.7-0.8)	37
Very good (0.8-0.9)	50
Excellent (0.9-1)	2

Vegetation density was the most common predictor, appearing in 38% of the models. This was followed by slope and percent ravine (both 31%) and catchment area upstream (20 %). Percent sandstone and percent limestone were the weakest predictors, only appearing in 10% and 8%, respectively.

When calculating the minimum sets, the summed rarity algorithm was slightly more efficient than the greedy heuristic and much more efficient than the maximum rarity algorithm (Table 5.4). The greedy heuristic algorithm required three more steps than the summed rarity algorithm, while the progressive rarity heuristic used 12 more steps. Although the difference in percentage of total area (Table 5.4) might seem small, the 0.5% difference in protected area between the best two algorithms equates to 1000 km^2. The remaining results are reported only for the summed rarity algorithm.

Table 5.4. Steps required for a minimum set using the thee different algorithms

Algorithm	Steps required	Percentage of total area
Greedy heuristic	35	10.6 %
Progressive rarity	44	11.3%
Summed rarity	32	10.1 %

When examining the selection patterns, it becomes obvious that only relatively small areas have to be protected to cover a large amount of taxa. Protection of only 2% of the area will protect 90% of the taxa in the dataset (Figure 5.2, Figure 5.3). Only the last 10% of the taxa require an additional reserve area of 8%. In other words, four times more area is needed to protect the final 10% of the taxa than the previous 90% (Figure 5.2).

The first steps of the heuristic algorithm reveal an interesting pattern (Figure 5.2, Figure 5.3). First small catchments with many rare taxa are selected. Then slightly bigger catchments are chosen, still with a bias towards rare taxa, but with diminishing returns (Figure 5.3). In step eight, the first catchment with predominantly common taxa is selected, adding 32 taxa to the protected list. The following stages consist of a mix of large and small catchments with rare and common taxa before the last ten steps select very large catchments that are needed to protect the remaining taxa. These catchments are the light coloured catchments in the centre and east of Figure 5.4.

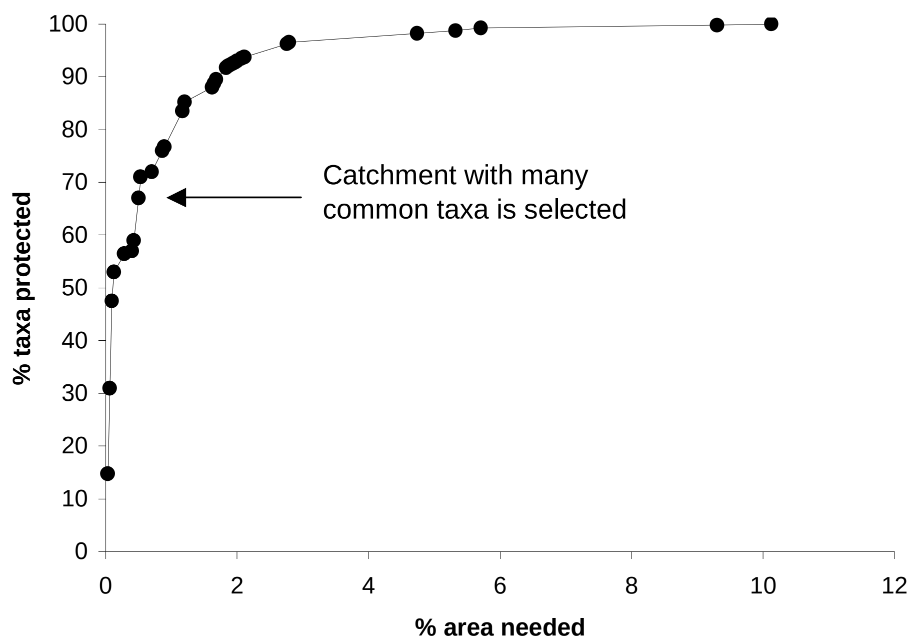


Figure 5.2. Returns curve for the summed rarity algorithm

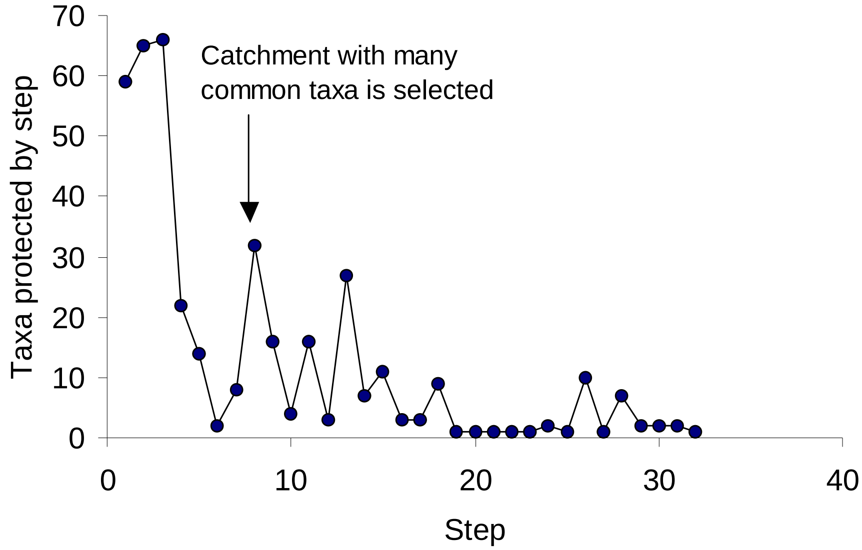


Figure 5.3. Number of taxa selected in each step of the summed rarity algorithm

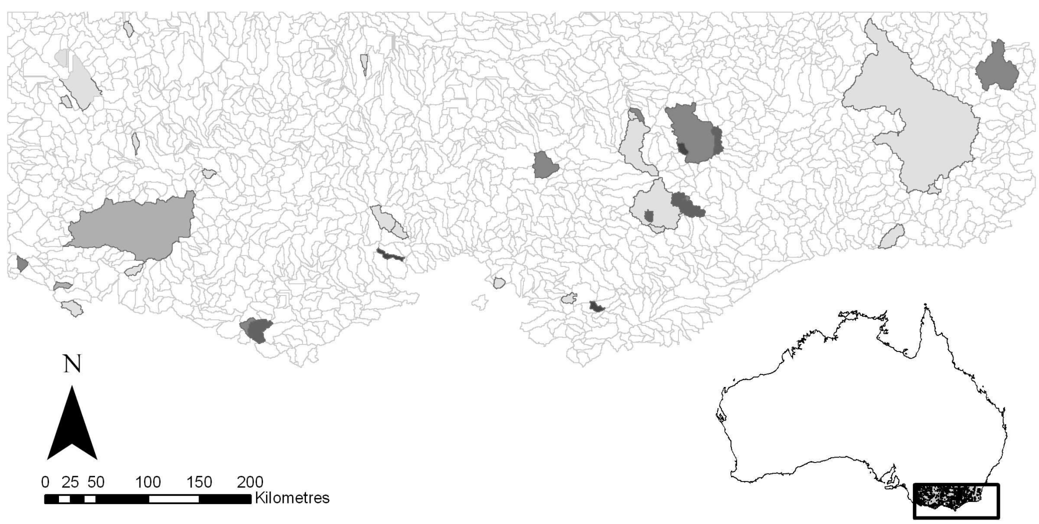


Figure 5.4. Minimum set from the progressive rarity algorithm. Darker colours indicate larger numbers of species protected by the selection step. Note that large catchments in the centre and east were only needed to protect 2 taxa each in the final steps.

Figure 5.5. Minimum set without the catchment restriction. Darker colours indicate a larger number of species protected by the selection step.

As a comparison, we ran the same algorithm without the catchment restriction (Figure 5.5). Only 1 percent of the total area was required in this case, differing from the whole-catchment-algorithm by a factor of ten. Many of the chosen subcatchments are in fact only the outflows of the larger catchments, selected in Figure 5.4. This demonstrates how introducing a catchment rule will change the planning scenario compared to non-connected systems.

When calculating irreplaceability, we started with removing 50 percent of the catchments randomly. After 1000 repetitions, only 370 catchments had been selected in one or more sets and therefore had non-zero values of irreplaceability. We then recalculated irreplaceability by removing 70% and 90% of catchments randomly. This increased numbers of catchments with non-zero irreplaceabilities to 558 and 908, respectively. Irreplaceability values with 50% and 90% removals are comparable (Figure 5.6). The top 100 irreplaceable catchments are identical in both solutions.

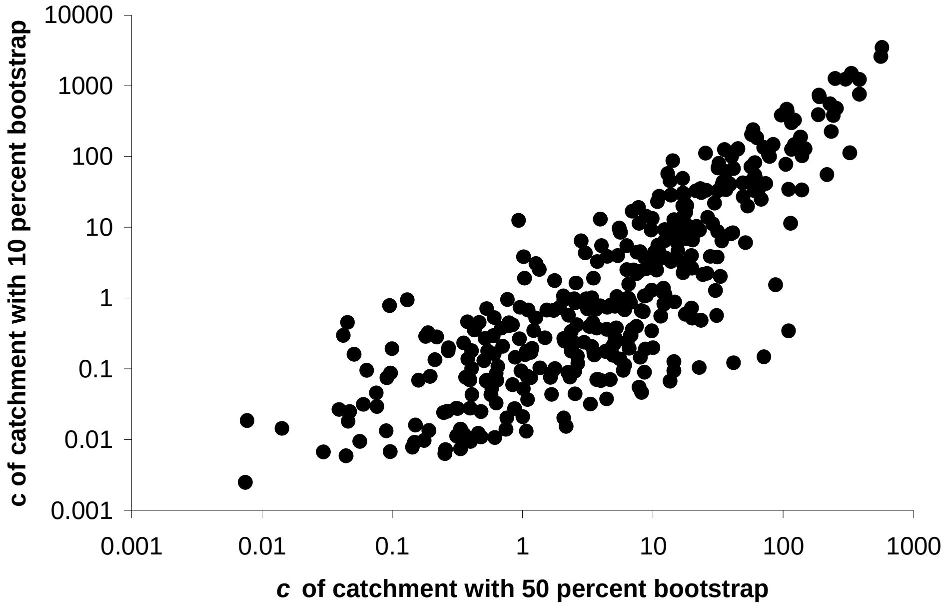


Figure 5.6. Correlations between catchment contributions (*summed c*) with different percentages of catchments taken out

A comparison of the three different algorithms using the 90 percent bootstrap demonstrates that the summed rarity algorithm again performs better than the other two algorithms, with only 24.7 steps required on average, compared to 30.4 and 33.2 respectively. Note that the average number of steps is less than the minimum set (Table 5.4), because not all taxa represented in the full dataset are present in the bootstrapped versions.

Table 5.5. Steps required in 1000 runs of the 90% bootstrap for three different heuristic algorithms.

	Steps required		
Algorithm	Minimum	Average	Maximum
Greedy heuristic	25	30.4	37
Progressive rarity	25	33.2	51
Summed rarity	19	24.7	32

A closer look at the data reveals more about the effect of the catchment restriction to estimate irreplaceability. For example, subcatchments 1789 and 1788 are small, adjacent headwater catchments in the southeastern region of Gippsland. Their irreplaceabilities are ranked 2 and 4 out of all catchments, respectively. Both predicted assemblages are almost identical but subcatchment 1789 is smaller (65km^2) than subcatchment 1788 (80km^2). This leads to catchment 1789 to be chosen first in the minimum set and, when estimating irreplaceability, in all cases where both catchments remain in the dataset. In these cases, catchment 1788 was not chosen in later steps, because its taxa - which are identical to those in 1789 - were taken out of the dataset to ensure complementarity. However, if 1789 was not present in the dataset, 1788 was always chosen and is therefore ranked highly irreplaceable. This creates our different groups of catchments:

1. Catchments that are always chosen when they are in the dataset (high *f*) and contribute many taxa (high average *c*, like catchment 1789)
2. Catchments that are not always chosen when they are in the dataset (mid to low *f*), but contribute highly when they are (high average *c*, like 1788)
3. Catchments that are always chosen but do not contribute many taxa (high *f*, low *c*)
4. Catchments that are occasionally chosen to provide a few additional taxa (low *f*, low *c*)

We found that 84 catchments were present in over 80% of solutions. Of these, 33 catchments contributed highly in most cases (Quadrant 1), whereas 51 contributed few taxa (Quadrant 2). Many of the latter catchments were fairly large with a few rare taxa. Twenty-five catchments were present in less than 80% of the solutions (Quadrant 3), but ranked highly if they were chosen. The remaining catchments 799 were chosen less frequently and did not contribute significantly (Quadrant 4).

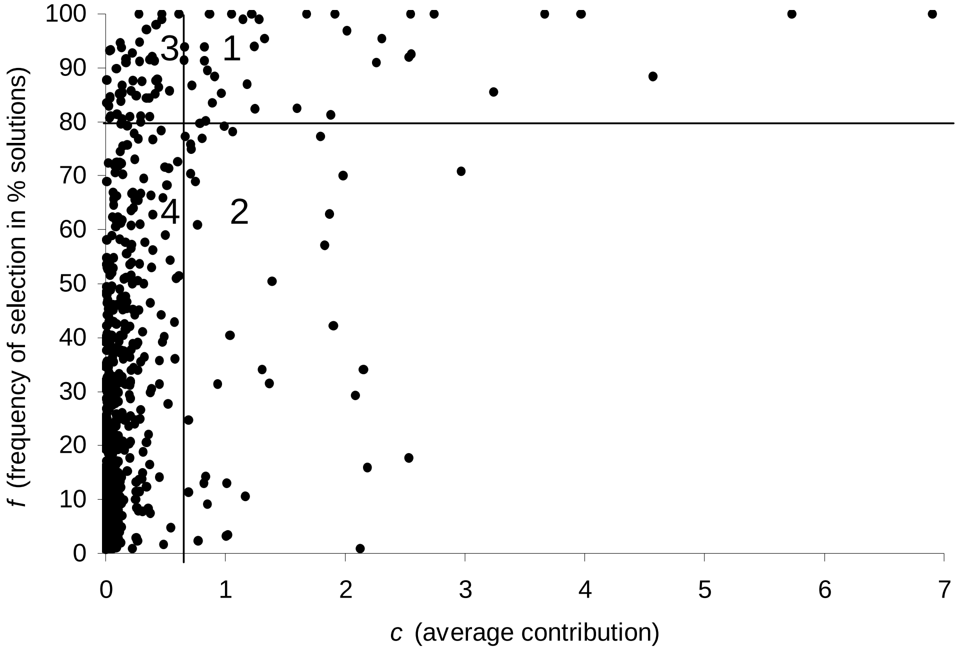


Figure 5.7. Categorization of the 908 catchments present in more than one solution. Quadrants refer to the groups of catchments discussed above.

When summing the contributions over the 1000 bootstrap runs, we get an estimate of irreplaceability that considers both frequency and contribution. It is strongly correlated to *c* (r^2=0.82) and weaker to *f* (r^2=0.27). A map of this measure for the study area reveals a strong bias for coastal and mountain lower order streams (Figure 5.8).

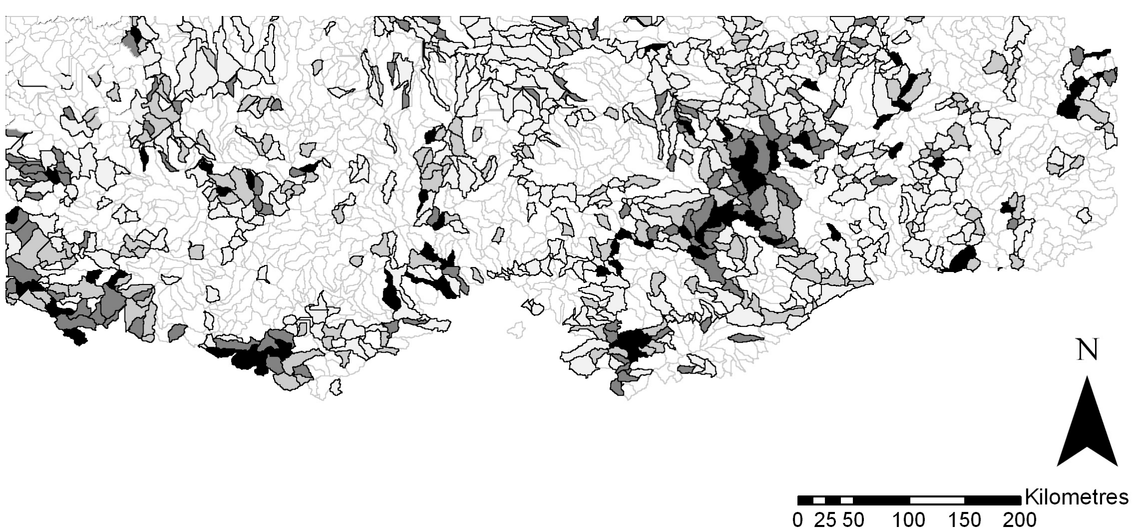


Figure 5.8. Irreplaceability map for the study area

5.5. Discussion

5.5.1. Prediction of invertebrates

When reducing the predictor variables to six PC axes to avoid overfitting, the first two axes represented the main gradients (Table 5.2). Axis 1 summarised local topographic variation and, with Axis 2 (upstream catchment area), reflected the position of subcatchments in their larger catchments. Axis 2 illustrates how seemingly independent variables are highly correlated in Victoria. Rainfall, latitude and position in the catchment are correlated, as often the higher parts of catchments are located in the north. All major groups of predictors were represented in the first six axes. Location descriptors were present in axes 1, 2 and 3, climatic descriptors in axes 2 and 5, and landform descriptors in axes 2,3 and 4 (Table 5.2). Geological variables were included on four axes and vegetation density was present in axis 4. Considering that all major groups were covered on multiple axes and 71 percent of the total variation was explained by these six axes, we conclude that most information is retained by selecting six variables while reducing the risk of overfitting (Makarenkov & Legendre, 2002; Yuan, 2004a).

Taxa modelling success was high, with 52 percent of taxa predicted at excellent or very good levels according to the classification by Boyd *et al.* (2005). Only 52 taxa (11.5%) could not be predicted. Surprisingly, upstream catchment area was not among the most important predictors. Position in the catchment is the most commonly used predictor of macroinvertebrate assemblages in predictive bioassessment programs and can be found in virtually every study using RIVPACS-style models (Wright, 1995; Hawkins *et al.*, 2000; Simpson & Norris, 2000; Ostermiller & Hawkins, 2004), approaches predicting metrics (Bailey *et al.*, 1998; Linke *et al.*, 1999), or predictive approaches using GAMs (Yuan, 2004a; Yuan, 2004b). It is relatively simple to derive distance from source or upstream catchment area, making them attractive choices of predictor variables that are also consistent with ecological theory (Vannote *et al.*, 1980; Minshall *et al.*, 1985). However, our results indicate that local predictors are more important than positional variables in the study area. We suspect that this related to the diversity of ecotypes in the study area: Taxonomic composition in the semi-arid western regions will be different from taxonomic composition in the montane climate of the east, even if they share the same relative position in their respective catchments.

The success of the predictions can be partly attributed to an unusual group of predictor variables: local variables that were not measured on ground, but derived from a GIS system. While a combination of variables is used in most bioassessment programs, we needed to extrapolate to unsampled catchments and were therefore restricted to metrics derived remotely and analysed in GIS. Programs like RIVPACS and AUSRIVAS are usually restricted to predictors that agencies and consultants can measure or derive easily, for example site-based measurements or basic information about position in the catchment. However, landform predictors from IDRISI and detailed vegetation and climatic descriptors proved to be important. Based on the high AUC for the invertebrate models, we suspect that these predictors are not only adequate replacements for on-ground observations, but might also enhance prediction in general. One possible reason is a more objective approach to obtaining local information without being subject to observer variation (Hannaford *et al.*, 1997; Poole *et al.*, 1997). Another reason for the success of large scale GIS variables is that the subcatchment represents a slightly different local scale than the actual site (for the influence of multiple scales on invertebrate community composition see Parsons *et al.*, 2003; Parsons *et al.*, 2004b).

The AUC values discussed above will have to be interpreted with a bit of caution, because no validation dataset was withheld (Yuan, 2004a). However, we do not consider slight overfitting as problematic for the purposes of our study, because it is not the goal of the modelling exercise to accurately predict the presence single taxa in a catchment. Instead, the models are used to produce a more informed stratification and weighting of environmental surrogates, similar to the techniques described by Sarkar et al. (2005). Instead of using bioregionalisation or artificially categorised habitat variables as representation targets we use large-scale environmental surrogates via taxa predictions. (Figure 5.9). Philosophically, this is similar to establishing an environmental dissimilarity relation sensu Faith (2003).

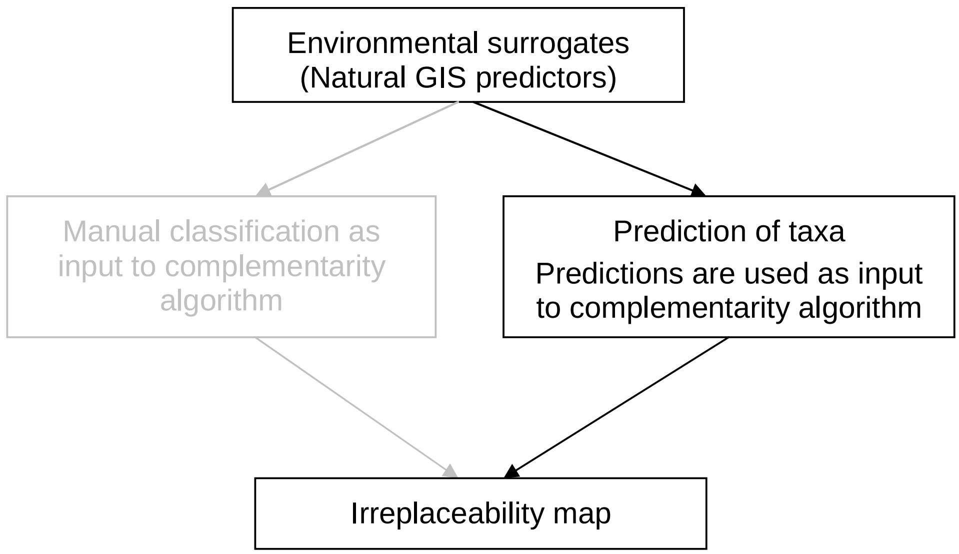


Figure 5.9. Two ways of using environmental surrogates to estimate irreplaceability. We stratified environmental surrogates by first predicting taxa.

5.5.2. Minimum sets and choice of algorithms

The summed rarity algorithm produced the most efficient solutions, both in the minimum set (Table 5.4) and the bootstrapped runs (Table 5.5). This is initially surprising, considering the progressive rarity algorithm was most efficient in the comparison by Csuti *et al.* (1997) and is implemented in ResNet (Kelley *et al.*, 2002; Sarkar *et al.*, 2002), as well as being in most other studies calculating minimum sets (examples are Lombard *et al.*, 1997; Araujo & Williams, 2000; Hopkinson *et al.*, 2000; Moore *et al.*, 2003). However, there is one fundamental difference between these studies and ours: many previous approaches have used planning units that are equal-sized grids, while we were dealing with catchments of different sizes, varying by a factor of 1000 from headwater subcatchments to the mouth of the Snowy River. Not only has the number of steps to be optimised, but also the catchment area.

An unmodified progressive rarity algorithm selects the catchment with the rarest feature. This potentially results in a very large selected catchment, requiring a correction for area. However, it does not make sense to correct the rarity of a single organism for area, leading to a relatively meaningless figure (Equation 2). In contrast, the greedy heuristic algorithm (Equation 1) selects for large numbers of taxa per unit area, a concept intuitively more appealing. The far greater efficiency of both algorithms that involved sums (i.e. richness and summed rarity) demonstrates that in a case where planning units vary greatly in size, both a measure of richness and area lead to relatively efficient networks. Considering the consistently better performance of the summed rarity algorithm (Table 5.5), it seems that the combination of richness and a rarity weighting with area can give efficient results for riverine planning and in other situations where planning units vary widely in area.

Efficiency is also illustrated by the taxon accumulation curves (Figure 5.2 and Figure 5.3). After selecting for small catchments with many rare taxa in the first few steps, there is a diminishing raw taxon count involving rare taxa until step 8 (marked by the arrow in Figure 5.2 and Figure 5.3). In this step, many common taxa in a small headwater catchment are selected, giving maximum return for a small area. The following steps alternate between selecting for numbers of taxa and rarity.

Although the differences in total area between the minimum sets and bootstrapped analyses might seem small at first glance, the 0.5 % difference between the two best algorithms, at land prices of A$1500-8000 ($US 1000-6000) per hectare, this would translate to between $15 million and $80 million. The difference between the best and the worst performing algorithms (1.5 %) translates to three times this amount. This demonstrates that even small differences in algorithm efficiency can make a large difference once the study area comprises an entire state.

After modifying the algorithm to fit the problem, the species accumulation curve for the minimum set looked similar to those observed in terrestrial studies (Vane-Wright *et al.*, 1991; Csuti *et al.*, 1997; Williams *et al.*, 2003). Characterised by a steep ascent at first, these accumulation curves typically deliver diminishing returns towards with later selections. In our case, the ascent is particularly steep and the diminishing returns very small. Only 2% of the total area is needed to represent 90% of the 400 taxa. The remaining forty taxa bring the required area to 10.3% of the total. Diminishing returns are exaggerated by the whole-of-catchment rule, where all of the upstream catchment must be protected. Some taxa are only found in lowland rivers with large upstream areas. Two very large catchments, shown in Figure 5.4 had to be added to cover these taxa.

Despite the whole-of-catchment restriction, the area required to meet protection targets is realistic – not the entire state will have to be protected to cover all taxa. In particular, the relatively small area needed to cover most taxa is encouraging and will hopefully lend support to calls for freshwater reserves (see Cullen, 2003; Dunn, 2003; Fitzsimons & Robertson, 2005; Kingsford *et al.*, 2005).

5.5.3. Irreplaceability

Our study presents the first estimate of irreplaceability that incorporates connectivity of river systems. While not as elegant as the irreplaceability coefficient by Ferrier *et al.* (2000), a bootstrapped selection algorithm has two advantages: First, estimation by bootstrapping can basically accommodate any underlying heuristic with any given set of rules. It therefore works with the whole-of catchment rule and will work with potential future developments that set flexible rules for partial catchment protection (see below). Second, withholding catchments from a potential solution emulates real- life management situations. The outputs of a bootstrapped irreplaceability algorithm are therefore easily understandable: *f* is the percentage of times that a catchment appears in solutions derived from data sets in which it was not removed. This is a measure of how essential a planning unit is to fulfil all targets. The other output is the *c*, the average contribution that a catchment made to the overall solution - a measure of relative importance.

Both of these properties contain valuable information, as the example of the two neighbouring catchments (1788 and 1789) shows. While both have an almost equally high contribution when they are selected, one catchment is only needed if the other catchment is unavailable, dropping its selection rate and irreplaceability value. A hypothetical opposite example are two catchments containing unique taxa. If catchment A had one unique taxon, but catchment B had five, both would have a selection rate of 100% but the summed contribution to targets of catchment B would be five times greater. Depending on the question, in a planning scenario one measure might be more informative than the other.

The frequency of selection is crucial when completeness of target representation is required: In this case, even catchments with low *c*, but high *f* will have to be selected. If maximum returns are to be achieved with only a few catchments, high *c* is equally important. Figure 5.7 demonstrates how the two measures are not necessarily linked, but also not completely independent. Catchments with the highest contributions will also have the highest frequency of selection – whenever they remain in the dataset, they will be selected first. As contributions decline, the measures become increasingly de-coupled. For the purpose of our study, we combine both by summing the contributions over the number of selections. Correlated to both *c* and *f*, we chose *summed c* as our measure of irreplaceability. *Summed c* is analogous to ‘summed irreplaceability across multiple features’, a statistical estimator described by Ferrier *et al.* (2000).

We found that 33 catchments were present in almost every solution, each contributing a large number of taxa. These catchments are the focus catchments for conservation action. Although there can still be some redundancy (see catchments 1788 and 1789), most of these catchments are highly important to achieve representation targets. For all representation targets to be achieved, another 58 catchments with lower contributions but present in almost all solutions would have to be considered. The remaining catchments are potentially valuable as replacements if highly irreplaceable catchments are unavailable.

The spatial distribution of irreplaceability makes intuitive sense. Highly irreplaceable catchments are found in every ecotype, from the dry western catchments to coastal temperate rainforest and the mountain regions. This is expected, because different taxa will be found exclusively in some of the ecotypes. There is a strong bias towards coastal catchments and inland headwater streams. The whole-catchment rule is responsible, making lowland river catchments less likely to be selected in the algorithm because of their large size. While there is some concern about this among conservation practitioners (Victorian EPA, pers. comm.), it is a fact that smaller headwater catchments will conserve more taxa per unit area, mainly because of the whole-catchment protection rules. Species-area effects (studied in the benthic environment by Taylor, 1996; Changeux, 1998; Woodward & Hildrew, 2002) are negligible, as Marchant *et al.* (in Press) found no strong relation between the size of the catchment and species richness for the same dataset. Modification of the catchment rule (see the following section) might be able to correct the bias towards headwater catchments and enable more efficient protection of lowland streams.

The computational effort is not as streamlined as the statistical estimator by Ferrier *et al.* (2000). Planning officers will not be able to calculate changes to configuration in real-time planning sessions. On the other hand, the algorithm is not as resource-intensive as the solutions by Tsuji & Tsubaki (2004) and can potentially handle very large datasets. Although the relatively clumsy program took a few hours to run in SAS (SAS, 2005), we would expect that an optimised version coded in a faster language would perform 1000 runs of 30 selection steps in about 10 minutes.

5.6. Conclusion and scope for future research

This study illustrates the value of an approach to river conservation that does justice to the spatial configuration of connected river networks. Figures 5.4 and 5.5 demonstrate the difference in outcomes with and without whole-catchment protection. Figure 5.4, the solution without the whole-of-catchment rule, is analogous to the state of protection of many rivers today - rivers that were coincidentally located in terrestrial reserves. Even if representation targets in rivers coincided with terrestrial targets, a fact disputed by Nilsson & Gotmark (1992), upstream disturbance could potentially render the reserve useless (Angermeier & Winston, 1999; Filipe *et al.*, 2004).

While the whole-of-catchment rule is a step in the right direction, future applications could make two modifications to the system. Pringle (2001) suggests upstream protection is necessary but not sufficient. Lateral cross-catchment disturbance can play a role as well as downstream connectivity (see also Yates & Bailey, in press-a for a recent example). Additional rules similar to the whole-of-catchment rule may have to be created to accommodate these threats.

The second modification works in exactly the opposite way. Depending on the stressor, the whole-of-catchment rule could be eased to recognise that not all stressors have an equal effect on downstream ecosystems. For example, mild organic pollution might be metabolised within 10 kilometres (Schwoerbel, 1972; often much less, Storey & Cowley, 1997) and recreational fishing could be allowed outside a core sanctuary. In these cases, protecting several links up- and downstream of an irreplaceable subcatchment would be sufficient. On the other extreme, sedimentation potentially influences the entire area downstream, as does over-extraction of water. In these cases, whole-catchment protection or at least control of the pollutant might be recommended. Mixed protection schemes where statutory reserves go hand in hand with community efforts (Lindenmayer *et al.*, 2000; Cowling *et al.*, 2003a), will be needed to achieve full protection.

Two further challenges to operationalise a complementarity-based assessment of conservation value are a) setting realistic conservation targets and b) embedding the irreplaceability coefficient in an overarching framework similar to the one described by Margules & Pressey (2000). Target setting is crucial to the success of a conservation plan. For demonstration purposes, we used one occurrence per taxon as a representation target. To ensure persistence (Gaston et al., 2002), especially of migratory taxa like fish, local managers and policy makers should be involved in a target setting process. Yet conservation value is only one part of the planning process. To prioritise conservation efforts where they really matter, both condition (Linke & Norris, 2003) and vulnerability (Margules & Pressey, 2000) will have to be considered.

For the present, the method presented here satisfies the three requirements we set out to achieve. First, it is based on complementarity, similar to the leading techniques in terrestrial and marine conservation planning and thus achieves high efficiency in achieving conservation targets. Second, the whole-of-catchment rule promotes complete protection when designing river reserves. However, our estimator of irreplaceability is also flexible to modification to accommodate planning scenarios in lowland rivers. Third, instead of the environmental surrogates used in other river planning exercises, our method relates directly to the biota. These three requirements set up the riverine irreplaceability index as the first step to systematic conservation planning in freshwater systems.

Chapter 6. Management options for river conservation planning: Condition and conservation re-visited

This chapter has been accepted for publication in the journal 'Freshwater Biology' (August, 2nd 2006)

S. Linke, R.L. Pressey, R.C. Bailey, R.H. Norris (in press). Biodiversity: bridging the gap between condition and conservation. *Freshwater Biology*.

6.1. Summary

1. Systematic conservation planning is a process widely used in terrestrial and marine environments. A principal goal is to establish a network of protected areas representing the full variety of species or ecosystems. We suggest considering three key attributes of a catchment when planning for aquatic conservation: irreplaceability, condition and vulnerability.

2. Based on observed and modelled distributions of 367 invertebrates in the Australian state of Victoria, conservation value was measured by calculating an irreplaceability coefficient for 1854 subcatchments. Irreplaceability indicates the likelihood of any subcatchment being needed to achieve conservation targets. We estimated it with a bootstrapped heuristic reserve design algorithm, which included upstream-downstream connectivity rules. The selection metric within the algorithm was total summed rarity, corrected for protected area.

3. Condition was estimated using a stressor gradient approach in which two classes of GIS Layers were summarised using principal components analysis (PCA). The first class was disturbance measures such as nutrient and sediment budgets, salinisation and weed cover. The second class was land use layers, including classes of forestry, agricultural and urban use. The main gradient - explaining 56% of the variation - could be characterised as agricultural disturbance. 75% of the study area was classified as disturbed.

4. Our definition of vulnerability was the likelihood of a catchment being exposed to a land use that degrades its condition. This was estimated by comparing land capability and current land use. If land was capable of supporting a land use that would have a

more degrading effect on a river than its current tenure, it was classified vulnerable (66% of the study area). 79% of catchments contained more then 50% vulnerable land.

5. When integrating the three measures, two major groups of catchments requiring urgent conservation measures were identified. Seven percent of catchments were highly irreplaceable, highly vulnerable but in degraded condition. These catchments were flagged for restoration. While most highly irreplaceable catchments in good condition were already protected, 2.5% of catchments in this category are on vulnerable land. These are priority areas for assigning river reserves.

6.2. Introduction

Australia's freshwater biodiversity is internationally recognized as highly significant (Dunn, 2003), partly for the degree of endemism of faunal groups and microorganisms (Groombridge & Jenkins, 1998) but also for high diversity in response to climatic variability and geological history (Lake, 1995; Schofield *et al.*, 2000; Dunn, 2003). However, a recent continental assessment showed that 86 percent of the rivers in an area of 3 000 000 km^2 were moderately to substantially modified (Norris et al., in press). Despite increasing degradation of freshwater systems and prominent scientists advocating conservation of river systems (Cullen, 2003; Kingsford *et al.*, 2005), biodiversity scientists have so far paid little attention to rivers (Cullen & Lake, 1995). Quoting Shakespeare, British limnologist Brian Moss - who calls for the catchment itself to be the focus of 'wise' and encompassing management – describes current conservation approaches as *sans teeth* (Moss, 1999).

The first priority in the recent Australian freshwater conservation literature is the assessment of conservation value and protection in the form of CAR (comprehensive, adequate and representative) freshwater reserves (Dunn, 2003; Fitzsimons & Robertson, 2005; Kingsford *et al.*, 2005). Cullen (2003) also recommends identification of threatening processes and protection against these, as well as restoration efforts to re-establish natural communities, a motion seconded by Everard & Powell (2002). The latter authors also highlight the need to consider whole catchments in the planning process.

Considering the above measures, we suggest a formal framework for integrated conservation management of rivers that should include three basic questions:

- Irreplaceability: What is the conservation value of the river and its catchment – and the organisms and habitats within?
- Condition: What is the condition of the catchment?

- Vulnerability: How is the condition likely to change without intervention?

These questions are far from new, although a management strategy that combines all three is a new contribution. Irreplaceability, vulnerability and their interactions are widely discussed in the terrestrial systematic conservation literature. Irreplaceability is commonly defined as the extent to which the loss of an area will compromise regional conservation targets (Pressey et al., 1994). Some areas will have many replacements to meet regional targets – others will have few or none. The latter are termed highly irreplaceable. The computational strategy of estimating irreplaceability is linked to the principle of complementarity. Complementarity-based selection algorithms in conservation theory look for areas that add as many under-represented surrogates (freshwater taxa in this case) as possible to a network of protected areas (Pressey *et al.*, 1997; Faith & Walker, 2002; Justus & Sarkar, 2002). In a freshwater context, assessments of conservation value have mainly been carried out based on fish (Angermeier & Winston, 1999; Filipe *et al.*, 2004) and river typology (Boon *et al.*, 2002; Fitzsimons & Robertson, 2005). We will use a new method for estimating the irreplaceability of subcatchments based on macroinvertebrates as the target organisms and recognising the need for whole-of-catchment connectivity for effective conservation of rivers.

Vulnerability - the possibility that future condition will change in a negative direction (Bradley & Smith, 2004) - has also been extensively studied in a terrestrial context. While most authors focus on the vulnerability of areas or landscapes to loss of biodiversity, Benayas & de la Montana (2003) define it in a species context and sum up the single taxon vulnerabilities in their planning units. In the most comprehensive review of vulnerability to date, Wilson *et al.* (2005a) state that 'the vulnerability of areas or features can be defined in relation to one or more proximate threatening processes'. Hereby, a threatening process ''threatens or may threaten the survival, abundance or evolutionary development of a native species or ecological community'' (Commonwealth of Australia, 1992; Commonwealth of

Australia, 1999). Wilson *et al.* (2005) divide vulnerability into three separate properties: exposure, intensity and impact. In this study we will focus on exposure - the probability of a threat expanding to a previously unaffected area. Examples given by Pressey *et al.* (2001) and Rouget *et al.* (2003a) use land capability and statistical models to predict expansions of agricultural use.

Although the issue of condition has been neglected in terrestrial conservation studies, it has been the main focus of aquatic ecology over the last decades. Traditionally, assessments of freshwater condition were site-based approaches, describing either habitat condition or changes in biota. Benthic macroinvertebrates - used since the beginning of the 20th century (Kölkwitz & Marsson, 1909) - are still the predominant indicator of stream condition, mainly in the form of predictive models (Resh *et al.*, 1995; Barbour *et al.*, 2000; Ofenbock *et al.*, 2004). Other site-based approaches use fish (Ibarra *et al.*, 2003; Kennard *et al.*, 2005), diatoms (Chessman *et al.*, 1999; Fore & Grafe, 2002; Bate *et al.*, 2004) or habitat attributes (Bain & Stevenson, 1999; Muhar *et al.*, 2000; Parsons *et al.*, 2004a).

While these site-based approaches can inform local management, modern large-scale assessments are instrumental to a conservation planning framework. The great advance in the last decades, which enables the link to conservation planning, is the increased use of geographic information systems (GIS) to extrapolate from sampled sites across whole planning regions. Data layers for GIS-based large-scale assessments of catchment or river condition have been integrated using hierarchical (Norris et al., in press) or rules-based (Walker et al., 2006) approaches. The Stressor Gradient Approach (Bailey et al., in press) identifies orthogonal stressor gradients by summarising input layers in a Principal Components analysis without *a priori* rules or assumptions. We used this approach to estimate condition in our study area.

This study is the first to consider irreplaceability, condition, and vulnerability at the same time and to combine them into a framework for decisions about protection and restoration. Systematic conservation planning traditionally considers two aspects, highlighting areas that are both highly irreplaceable and highly vulnerable as priorities for conservation action (Pressey & Taffs, 2001; Noss *et al.*, 2002; Cowling *et al.*, 2003a). Both the approach by Linke & Norris (2003) and SERCON, the British system for river conservation (Boon *et al.*, 1998; Boon *et al.*, 2002) deal with conservation value and condition, but do not consider vulnerability. The Australian Wild Rivers project (Stein et al., 2002) only considered condition in the form of 'naturalness' for their approach to river conservation.

The first objective of this study was to develop workable estimators for irreplaceability, condition and vulnerability in the state of Victoria, Australia. We used benthic macroinvertebrates as our targeted features for measuring irreplaceability. For condition and vulnerability, we will use disturbance layers from Australia's First National Land and Water Audit (Norris *et al.*, 2001; Walker *et al.*, 2006) and data on land use and land capability. The second objective was to integrate the three measures into a common framework (ICV – irreplaceability, condition, vulnerability) and establish appropriate management responses for different combinations. Based on this framework, we will propose management recommendations for different combinations of measures and flag priority areas for conservation reserves and restoration measures.

6.3. Methods

6.3.1. Study area and data

Victoria - a state in south-east Australia covering roughly 227 600 km^2 - has a wide variety of landforms and climatic conditions, which makes it an ideal area for developing and testing new methods. Environments range from alpine areas of up to 2200 m altitude in the north-east to temperate rainforest along the coastlines and semi-arid regions in the western plains. We used most of the state in our study, excluding a small corner in the north-west, which was left out for two reasons: First, part of this corner has no water at all and second, it is part of a much larger that spans across three other states and has a distance to source of over 1000 km, which would make planning scenarios impossible to control and quantify at the state level.

Sixty-one percent of the study area is tenured as agricultural land, the vast majority of which is grazed at moderate to low intensity. This is mainly found in the western regions, while the 23 percent used for forestry is predominantly located in the eastern part. One percent is urbanized, mainly around Melbourne, the sprawling state capital. Twelve percent of the land is protected in the form of National and State Parks, nature reserves and other areas that meet the criteria of the World Conservation Union (IUCN, 1994).

Systematic conservation planning studies in terrestrial environments often use equal sized grid cells as planning units (Gaston & David, 1994; Williams *et al.*, 1996; van Jaarsveld *et al.*, 1998; Sarkar *et al.*, 2002; Moore *et al.*, 2003). As this is not appropriate for rivers, we chose subcatchments as our units of planning and management to recognise the connected nature of rivers and the natural boundaries of areas of influence. We derived 1854 subcatchments from a modified 3 arcsecond digital elevation model (DEM). Digital data was sourced from the NASA SRTM mission (van Zyl, 2001) and analysis was carried out using ARC Hydro (Maidment, 2002) and Hawth's Tools (Beyer, 2004) within ArcGIS 8.2 (ESRI, 2002). We used two spatial scales in the analysis: The subcatchment scale was used to model the distributions of macroinvertebrate taxa. To assess condition and vulnerability, we were guided by the principle that 'the valley rules the stream' (Hynes, 1975) and considered the compounding effects of upstream stressors (Allan *et al.*, 1997; Yates & Bailey, in press-b). Therefore when calculating the effects of disturbance and vulnerability, we considered the entire catchment area upstream of, and including, the subcatchment. This is the definition of 'catchment' as used in the subsequent analysis.

6.3.2. Irreplaceability

Irreplaceability was calculated using benthic macroinvertebrates as the targeted features. Invertebrate samples were collected in 222 subcatchments using the rapid bioassessment protocol described by Marchant *et al.* (1997) and Metzeling *et al.* (2003). Samples were live-picked in the field for 30 minutes until about 200 individuals were retrieved. These were identified to species level where possible, resulting in a total of 1065 taxa. Subcatchments are represented by a riffle (kicknet) and an edge sample (sweep-net) each, collected in two seasons (spring and autumn).

As a first step in the analysis, we used the records from the 222 subcatchments in reference condition (Reynoldson et al., 1997) to predict occurrences of taxa in other subcatchments. Lacking on-site observations for the unsampled catchments, we used GIS-derived predictors to build generalized additive models (GAMs, Hastie & Tibshirani, 1999) (GAMs, Hastie & Tibshirani, 1999) for single invertebrate taxa. The twenty-four predictor variables are described in more detail in Linke, Norris & Pressey (in prep.) and include descriptors from the following categories:

- Catchment metrics such as elevation, distance from source and distance to mouth
- Percentage of landform categories summarised by subcatchment – for example percent ridge or percent concave/convex slopes - derived from using the 'toposhape' module in IDRISI Kilimanjaro (Clark Labs, 2004)
- Average and variation of rainfall and temperature by catchment and subcatchment
- Vegetation categories (Barson et al., 2000) and geology (Palfreyman et al., 1976) for whole catchments

When constructing the GAMs, two measures were taken to avoid overfitting. First, taxa with less than ten recorded observations in the dataset were omitted from further analysis. Second, predictors were analysed using principal components analysis (PCA) in S-Plus (Insightful, 2005) to eliminate correlated variables (Yuan, 2004a). We kept predictors with the highest loadings on the first six principal components (analogous to Yuan, 2004a). These predictors were:

- Slope in the subcatchment
- Rainfall
- Percent of convex slopes (calculated by IDRISI)
- Percent of sandstone in the catchment (from Palfreyman et al., 1976)
- Percent of limestone in the catchment (from Palfreyman et al., 1976)
- Vegetation density (from Barson et al., 2000)

Using a stepwise algorithm similar to the one described in Yuan (2004b), 400 taxa were successfully modelled at AUC>0.6 (Fielding & Bell, 1997). Probabilities of occurrence for these taxa were assigned to the unsampled subcatchments and then converted to presence/absence using a threshold of 0.5, similar to AUSRIVAS/ANNA (Simpson & Norris, 2000; Linke *et al.*, 2005).

For regional data sets, exact measurement of irreplaceability values is usually impossible, so estimation methods are used in planning (Jacobi et al., in press). For this study, irreplaceability values were estimated for each subcatchment using a bootstrapped heuristic complementarity algorithm. To recognise the connected nature of rivers, we introduced an additional rule that non-headwater subcatchments cannot be protected without subcatchments further upstream. Single subcatchments were flagged as forbidden configurations if they were not headwater catchments (Figure 6.1). This leads to two modifications to the traditional heuristic algorithms (Margules et al., 1988). First, instead of only considering taxa in single subcatchments, the subcatchments upstream of selected non-headwater subcatchments also have to be included in the selection, along with the taxa they contain. This group of subcatchments then constitutes a single entity for the purposes of the selection algorithm and the estimation of irreplaceability.

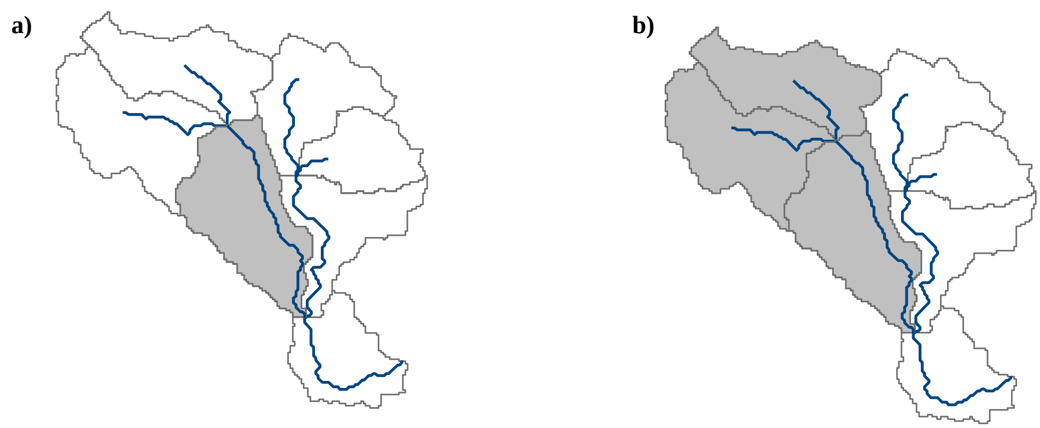


Figure 6.1. Illustration of forbidden and allowed configurations in the complementarity algorithm. Protected subcatchments are highlighted by shading. Isolated subcatchments that are not headwater catchments are forbidden configurations (a). Groups of subcatchments that include headwaters have to be protected instead (b)

Second, because selections consequently varied widely in extent, a measure of the number and rarity of taxa protected in each subcatchment or group of subcatchments (hereafter termed ‘contribution to targets’) has to be corrected for total area.

We therefore developed the following index:

$$c = \sum \frac{1}{f} / area \qquad \text{(Equation 1)}$$

where c = contribution to targets, summed across all taxa in the subcatchment or group of subcatchments, corrected for area

f = frequency of the taxon in the entire dataset

area = hectares covered by subcatchment or group of subcatchments

The first step of the algorithm was to select the subcatchment (or group of subcatchments) with the highest *c*. This subcatchment and the taxa it contained were then removed from the dataset and *c* was re-calculated. Tied values of *c* did not occur so we did not have to resort to tie-breaking rules (Pressey et al., 1997). Selections, removals, and recalculations were repeated until every taxon in the dataset was represented at least once.

This algorithm delivers a single, minimum or minimum configuration to protect all taxa. To estimate irreplaceability *sensu* Pressey *et al* (1994) and Margules, Pressey & Williams (2002), we needed to estimate the proportion of all possible minimum configurations in which each subcatchment was included. We ran a bootstrap analysis on the heuristic algorithm. The analysis was run 1000 times and a random 90 percent of subcatchments were marked unavailable for protection in each run (Linke et al., in prep.). If non-headwater subcatchments remained available for protection, so did their linked upstream subcatchments. Contributions to target were recalculated for each run, recognising the reduced frequencies of taxa in the remaining subcatchments. The algorithm was then used to select a representative set of subcatchments from the reduced number available. The final irreplaceability value was calculated by adding the contributions of each subcatchment across the 1000 runs. To categorise irreplaceability, we split the results into percentiles based on the sum of contributions. The highest category of irreplaceability is defined by the 90th percentile, further cut-offs are set at the 75th percentile and the median.

Only data from reference subcatchments were used to produce the distribution models, therefore some invertebrate taxa predicted to occur in unsampled subcatchments will not be present because of degraded environmental condition. Irreplaceability is therefore interpreted here as referring to the potential composition of subcatchments if pristine conditions were to be re-established and, for this exercise, assumes that the models are correct.

6.3.3. Condition

In the First National Land and Water Resources Audit (Norris et al., 2001) on-ground biological data were extrapolated across unsampled subcatchments in parts of the state to measure condition. We needed to answer the question ‘What is the current condition?’ for the whole state of Victoria. To ensure general applicability in large-scale planning scenarios without extensive on-ground data, we used an approach based exclusively on GIS data. The analysis, from now on referred to as the ‘stressor gradient approach’ (SGA, Bailey et al., in press) combines GIS layers using Principal Components Analysis (PCA) to identify orthogonal (uncorrelated and non-redundant) stressor gradients.

We selected stressor GIS layers that had well documented links with river condition in the literature. These can be summarised in two categories (Table 6.1). The first category includes direct measures of catchment condition, mainly taken from two themes of the First National Land and Water Resources Audit (NLWRA, Tait et al., 2000). Nutrient and sediment budgets were calculated by mapping the sources of sediment and nutrients to each subcatchment from hillslopes, gullies, riverbanks and major point sources. After subtracting the deposition of sediment in floodplains and lakes, and the additional losses of nitrogen to the atmosphere, these current annual loads were compared to modelled natural loads to calculate a deviation from the natural state (Norris et al., in press). Catchments with high soil acidification risk were defined as high intensity agricultural areas on soils with a low acid buffering capacity (Walker et al., 2006). Current extent of dryland salinisation and percent covers of native vegetation and weeds were also taken from the NLWRA assessment of catchment condition (Walker et al., 2006) .

Table 6.1. GIS layers used to define stressor gradients in Victoria

Catchment condition layers	Land use layers
Nutrient load	Intensive agriculture class
Sediment load	Road density
Acidification	Percentage of Land use classes
Salinity	• Grazing
Percent of native vegetation	• Forestry
Percent of weed invasion	• Tilled Agriculture
	• Urban
	• Conservation

Closer to the approach of Bailey *et al.* (in press), the second suite of GIS layers included indirect disturbance measures related to land use. Road density was estimated from the AUSLIG 250k national coverage. The percentage of land use in each subcatchment was estimated from Version 2 of the Interim Land use Map of Australia (Stewart et al., 2001). To simplify land use categorisation, we used the Audit Commodity Classifications (Stewart et al., 2001) for the stressor gradient analysis. Categories in our area of studies were Grazing, Tilled Agriculture, Forestry, Urban and Conservation. Classes of intensive agriculture (tilled agriculture on land with a high salinity risk) were taken from Walker *et al.* (2006). Degrading effects of land use categories on river condition are discussed extensively in the literature (Lenat & Crawford, 1994; Allan *et al.*, 1997; Boulton & Brock, 1999; Allan, 2004; Gourley & Ridley, 2005).

Road density and class of agricultural intensity were averaged by catchment, while land use categories were summarised as percent cover. A Principal Components Analysis (Pearson, 1901) was carried out in SAS (SAS, 2005) on the covariance matrix to reduce the stressor layers to three orthogonal gradients.

6.3.4. Vulnerability

While an ideal system to estimate the future vulnerability of a catchment would include exposure, impact and intensity (Wilson et al., 2005a), actual approaches (including ours) are usually limited to exposure because of the difficulty of deriving consistent data on intensity and impact across large areas. Our definition of vulnerability was the likelihood of a catchment being exposed to a land use that degrades its condition. To estimate this potential, we considered two factors – the current tenure and land use in subcatchments and the capability of subcatchments to support a different land use that would increase biodiversity loss. Notionally, our estimate reflects the likelihood of further decline in condition over a certain period. After this period, say five years, it will be important to have new estimates of vulnerability to deal with unforeseen events (Wilson *et al.* 2005).

We estimated agricultural land capability with a simplified version of the approach taken by Emery (1985) and Cunningham *et al.*(1998). The original eight land capability classes were grouped into three broader classes:

1. The highest capability class (analogous to Emery Class I-III), suitable for cultivation, was characterised by upper slopes less than 5 percent, erosion risk that is zero to moderate, and no excessive salt accumulation (<4 mS cm^{-1}).
2. Medium capability (analogous to Emery Class IV-VI) can support grazing and was characterized by upper slopes that average < 25 percent, but are always less than 33 percent. Erosion risk is moderate, but salt concentrations can be high.

3. Low capability land (Emery Class VII/VIII) was characterized by steep slopes and high erosion risk. This land is recommended to remain without cropping, grazing or forestry.

The key assumptions of our approach to estimate vulnerability are:

1. Higher-intensity land uses will, per unit area, contribute more to loss of river biodiversity. In accordance with the catchment disturbance index (CDI) in the NLWRA Assessment of River Condition (Norris *et al.*, 2001; Norris *et al.*, in press) we rank the influence of land use categories in the following order from low intensity to high intensity: protected areas, grazing on native vegetation, forestry, planted pastures, and tilled agriculture.

2. Appropriate land uses will occur in relation to land capability categories, for example only extensive grazing will occur on land not capable for intensive cultivation.

3. Subject to tenure, a greater proportion of remaining native vegetation on higher capability land will be converted to pasture or crops.

To simplify catchment integration, we ignored the magnitude of potential degradation, and only used a binary estimator (likely to worsen: yes/no) of potential vulnerability, summarized in Table 6.2. Protected public land is not classified as vulnerable, but vegetated mid- and high-capability public land is. This implies the potential for intensification of logging or a shift from grazing to logging or tilled agriculture. Note that most of the forestry activity in this area consists of non-clearfelling forestry in mixed-use forests. Therefore, an extensive literature review in the framework of the National Land and Water Audit (Norris et al., 2001) classified its effect on rivers as ‘medium disturbance’ - more than grazing on native pasture, but less than (mostly irrigated) cropping.

Similar threats exist for unreserved pastured land and vegetated private property. Private pastured land is only vulnerable to a switch to cropping if land capability is high. In this very coarse classification, we regarded intensively tilled agricultural land as not vulnerable, because it is the highest intensity land use. In future classifications, the possibility of urbanization or mineral exploitation as well as management practices could be considered, thus rendering tilled agricultural land as potentially vulnerable to degradation in condition.

Table 6.2. Method for estimating vulnerability to degradation. Shaded cells are vulnerable

Potential change in impact

Present impact class

	Potential for conversion to a higher intensity land use under land capability		
Land use	Low capability	Moderate capability	High capability
Strict reserve (vegetated by definition)	NO	NO	NO
Unassigned public or private land (vegetated)	NO	YES	YES
Private land (native pasture)	NO	YES	YES
Public or private land (Forestry)	NO	YES	YES
Private land (planted pasture)	NO	NO	YES
Private land (cultivated)	NO	NO	NO

The binary trajectory of vulnerability was calculated at the 9 arc second pixel scale (limited by the resolution of the land use database) and coded with 0 (no vulnerability) and 1 (potential vulnerability) respectively. Using Spatial Analyst in ArcGis 9 (ESRI, 2002) we calculated the percentage of land vulnerable to change for each catchment.

6.3.5. Management integration

The three assessments – irreplaceability, condition and vulnerability – can be represented as three axes (Figure 6.2), with a gradient of appropriate management responses attached to each. In this paper, we generalize these gradients according to “high” and “low” categories for each axis. High irreplaceability indicates few or no spatial options for protecting or restoring features relative to targets. Therefore these catchments should be the main focus of conservation action. Highly irreplaceable subcatchments in degraded condition will need restoration efforts. Here, we emphasise that we calculated a ‘potential irreplaceability’ based on predicted distributions of taxa that were extrapolated from catchments in good condition. We make the (optimistic) assumption that if the habitat in the catchment were restored to approximate natural conditions, the original biota would return. In subsequent work, the uncertainty behind this assumption can be explored. In this context, we define ‘restoration’ as the appropriate management response to degraded condition. Vulnerability, the likelihood or imminence of further loss of biodiversity in an area (Wilson et al., 2005a), requires protective measures, especially for subcatchments with high irreplaceability.

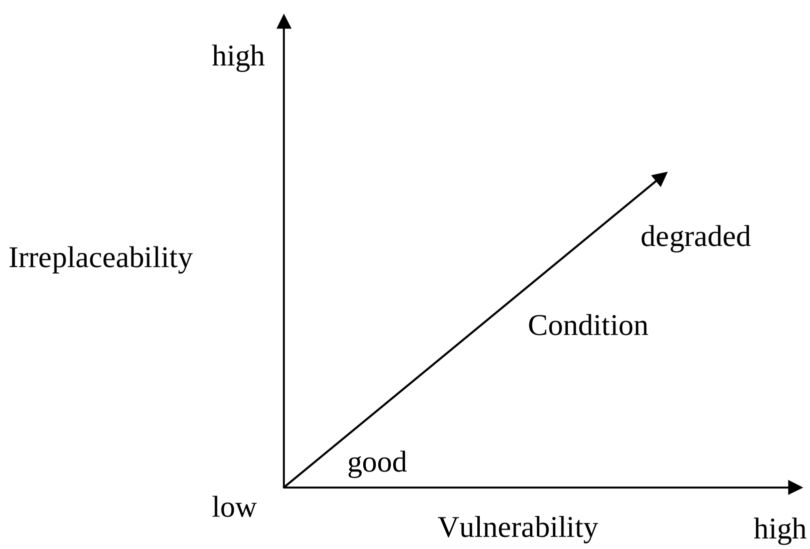


Figure 6.2. The three axes on which subcatchments lie in this study

Analogous to Margules & Pressey (2000), we divide our decision space into four quadrants that are defined by high and low values of condition and vulnerability. We use this to illustrate management responses only for subcatchments with high irreplaceability. The corresponding table for subcatchments with low irreplaceability is not presented here, but would emphasise responses that explore alternative spatial options for protecting or restoring particular subcatchments. It is important to note, however, that areas with low irreplaceability can contain taxa and other features of concern that are not present in areas with high irreplaceability. This means that a proportion of subcatchments with low irreplaceability will need to be protected and restored to achieve conservation targets (Margules and Pressey 2000; Pressey and Taffs 2001).

Table 6.3. Decision matrix integrating condition and vulnerability assessments for highly irreplaceable subcatchments

		Condition	
		Good	Degraded
Vulnerability	Low	1: Good condition and low risk of degradation. No immediate need for action but, especially in highly irreplaceable areas, vulnerability assessments should be updated regularly to monitor whether subcatchments have moved into the highly vulnerable category.	2: Poor condition and low risk of further degradation. These subcatchments are priorities for restoration. Protection is not needed in the short-term, but vulnerability assessments should be updated regularly.
	High	3: Good condition and high risk of degradation. These are priority areas for protection and listing as reserves. By definition, a large proportion of highly irreplaceable subcatchments will need to be protected if conservation targets for taxa are to be achieved.	4: Poor condition and high risk of further degradation. These subcatchments require a dual approach to management: (1) protection to offset land use changes that will cause further loss of condition; and (2) restoration to move condition back toward that of reference catchments.

Priority catchments for protective management are in quadrant 3 (Table 6.1). We recommend that these catchments are listed as aquatic reserves or other classifications with secure tenure. This follows the logic of Gaston, Pressey & Margules (2002), who argue that, in conservation science, protection is better than cure, especially given the uncertainties of restoration success and the extended lags in recolonisation of some species (Wilkins et al., 2003).

This principle should also guide managers when considering quadrant 1. Although there are no imminent threats, vulnerability assessments should be updated regularly and – in case of change – protective measures taken.

Areas in quadrant 2 are flagged for restoration efforts. In areas of high irreplaceability, re-colonization programs might be considered for rare and iconic taxa. These areas are not likely to degrade in the near future. However the watching brief required for quadrant 1 also applies here. For subcatchments with degraded condition and high vulnerability (quadrant 4), both protection and restoration are needed.

6.4. Results

6.4.1. Irreplaceability

Based on the 222 reference subcatchments, we successfully modelled the distributions of 400 taxa across the 1854 subcatchments. Despite setting the AUC threshold for successful models as low as 0.6 (Mason & Graham, 2002), most GAMs had AUC values between 0.7 and 0.9 – these indicate good predictions. Only 10 percent of the modelled taxa had AUC values between 0.6-0.7, which are defined as borderline acceptable.

908 subcatchments were chosen in at least one run of the heuristic algorithm. Most of the 90 subcatchments in the highest irreplaceability category were chosen in every randomisation in

which they occurred. Their total contribution to the overall targets $c = \sum \frac{1}{f} / area$ (see Equation 1) was four times higher than contribution of subcatchments in the remaining 90 percent. Many of them are headwater or coastal subcatchments distributed in all major environments, but concentrated in the central highlands. Subcatchments in the next highest category (between 90th - 75th percentile) are spatially associated with those in the 90th percentile.

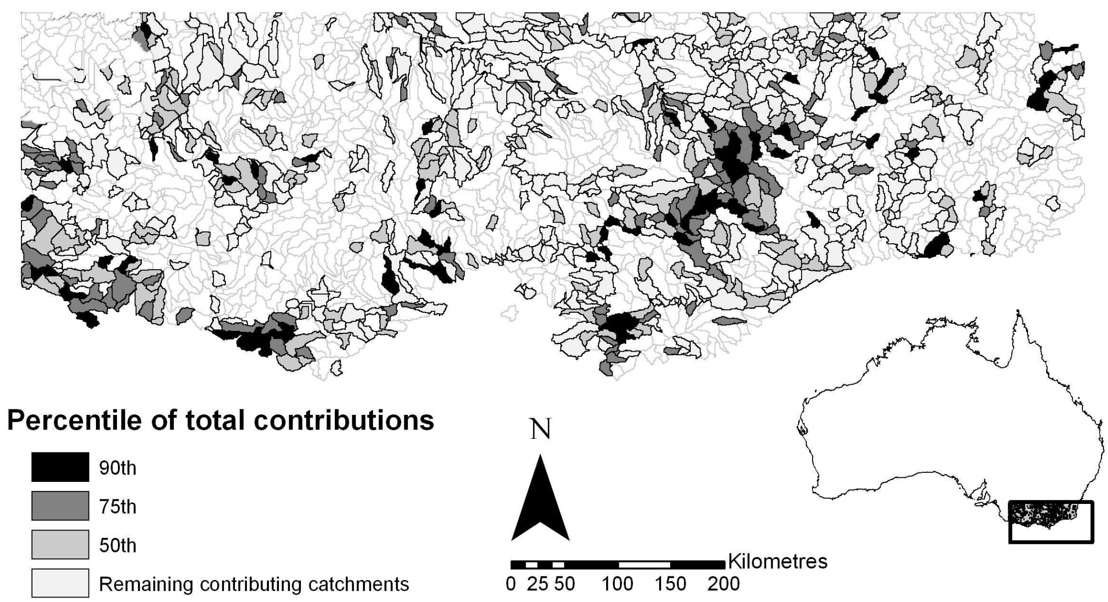


Figure 6.3. Classification of irreplaceability values in the study area by the percentile of contributions. Darker colours indicate higher irreplaceability

While many of these subcatchments occur in almost all randomisations, but with much lower contributions, some are only chosen a few times but represent a high proportion of the taxa list (large contributions) when they are chosen. This happens when very similar subcatchments that have a slight advantage in size or assemblage are made unavailable by the randomisations. In the category between the 75th and 50th percentile, most subcatchments are chosen regularly yet make smaller contributions. Subcatchments in the lower 50 percent only occur as replacement catchments and make small contributions to targets, while 950 other catchments are eliminated in the bootstrapping and do not contribute.

6.4.2. Condition

Three principal axes were identified in the principal components analysis of stressors, explaining cumulatively 75 percent of the variation in the data. The first stressor gradient explains 51 percent of the total variation (Figure 6.4). Characterized by both intensive agriculture and grazing, it can be described as an agricultural stressor gradient. The more proximate catchment condition measures associated with this gradient were increased sediment load, as well as high acidification risk. The high end of this stressor is mainly found in the highly agricultural western regions of the state, yet some impacted catchments are also found on the central Gippsland coast and in the highlands in the east of the state.

Table 6.4. PCA results to identify stressor gradients. Significant eigenvalues of stressor variables are in brackets.

PC 1: Agriculture (51% explained)	PC 2: Urban (14% explained)	PC 3: Urban/Forestry (12% explained)
Sediment load (0.36)	Nutrient load (0.51)	Nutrient load (-0.46)
Intensive Agriculture (0.41)	Impoundment density (0.39)	Road density (0.50)
Native Vegetation (-.42)	Road density (0.50)	Urban (0.38)
Acidification (0.37)	Urban (0.53)	Forestry (0.32)
Grazing (0.40)		
Forestry (- 0.40)		

The second gradient, explaining 14 percent of the variation can be described as an urban gradient. It is mainly centred around the capital city Melbourne (population 4 million) and is associated with an increased nutrient load (presumably from urban runoff) and high impoundment and road density. The third gradient accounts for another 12 percent and is associated with smaller urban centres and forestry. Because of the small spatial spread of the urban gradient and the low explanatory power and the comparably small impact of forestry (Norris et al., 2001), we will focus on the agricultural gradient in further analysis.

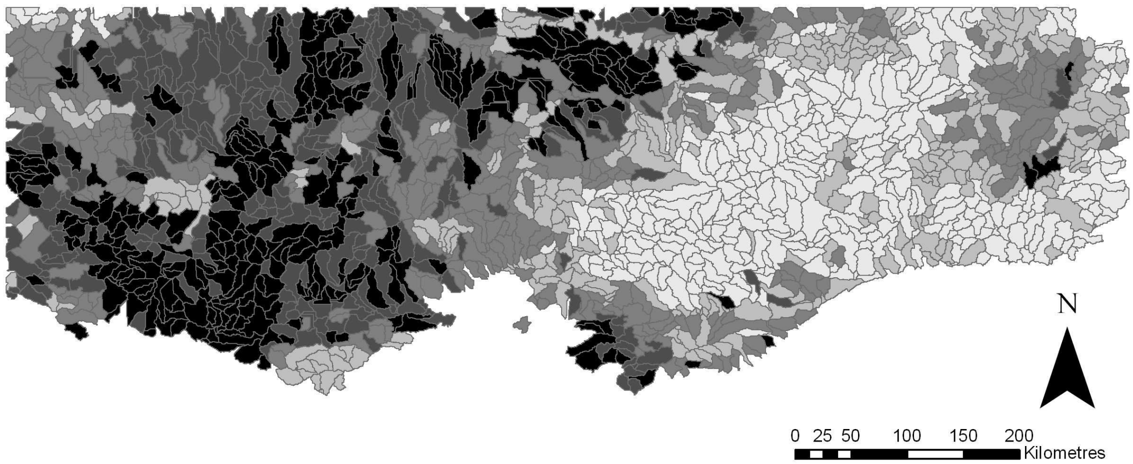


Figure 6.4. The agricultural stressor gradient (PC1) in the study area. Darker colour indicates a higher loading on the principal axis and therefore more degraded condition

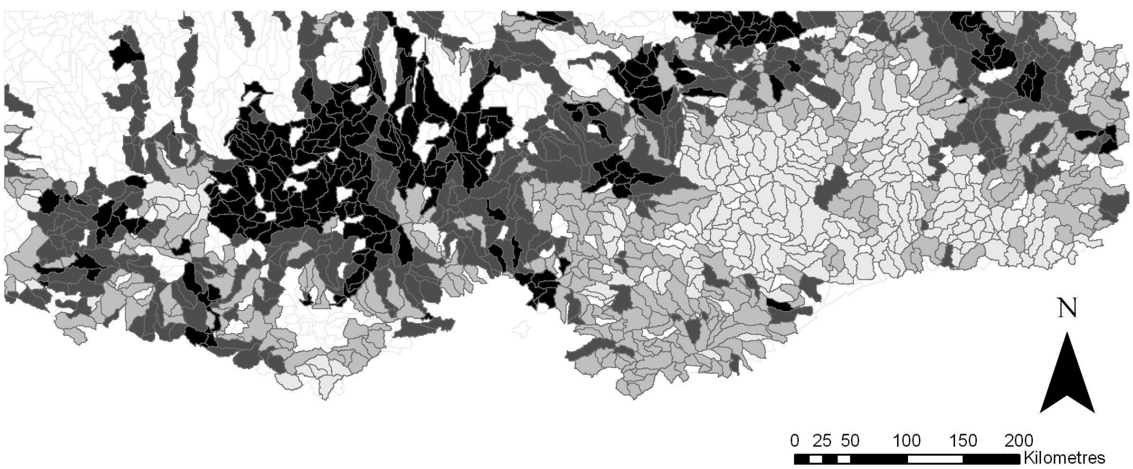


Figure 6.5. ARC_E from the Assessment of river condition (ARC). Darker colour indicates lower ARC_E and more degraded condition. White subcatchments were not assessed by the ARC.

As a validation, we compared the gradients with the scores of the Assessment of River Condition (Norris *et al.*, 2001; Norris *et al.*, in press). Although the significant correlation was not very strong (r^2=0.26), the patterns are very similar when at the large scale. Correlations with the other two gradients were non-significant. Both, the agricultural and the urban gradients were significantly correlated with AUSRIVAS O/E scores, a measure of site-specific taxa loss (Simpson & Norris, 2000; Wells *et al.*, 2002).

6.4.3. Vulnerability

63 percent of the area had a land capability of I-III and was therefore suitable for cropping. 32 percent of the area was classed as category IV-VI and therefore suitable for grazing and forestry, of which 7 percent was flat land with low soil erosion risk but some salinity risk. Only 5 percent - mainly steep slopes in the mountain regions - were assigned category VII-VIII (unsuitable for cropping or grazing).

66 percent of the land was classified as vulnerable to a decline in condition, measured at a cellsize of 0.01 decimal degrees. Native pasture as well as forestry on land suitable for sown pasture or cultivation represented the most abundant categories of vulnerable land (21 percent and 19 percent respectively) (Figure 6.6). Unallocated private property on Class I-IV land that could be used for pasture or cultivation made up 16 percent. The remaining 10 percent is land of the highest category, where sown pasture could potentially be converted to crops. Aggregated to subcatchments, 79 percent were classed as highly vulnerable (>0.5 of their areas with potential for intensification).

When estimating vulnerability, the 12 percent of the area that was already protected in conservation areas according to IUCN (1994) guidelines was flagged as not vulnerable. This is visually represented by lighter areas in the northeast of Figure 6.7. The other two categories that were automatically labelled non-vulnerable were the 8 percent of land already under cropping and the 1 percent of land classes as urban. The bulk of these areas are in the northwest of Figure 6.7.

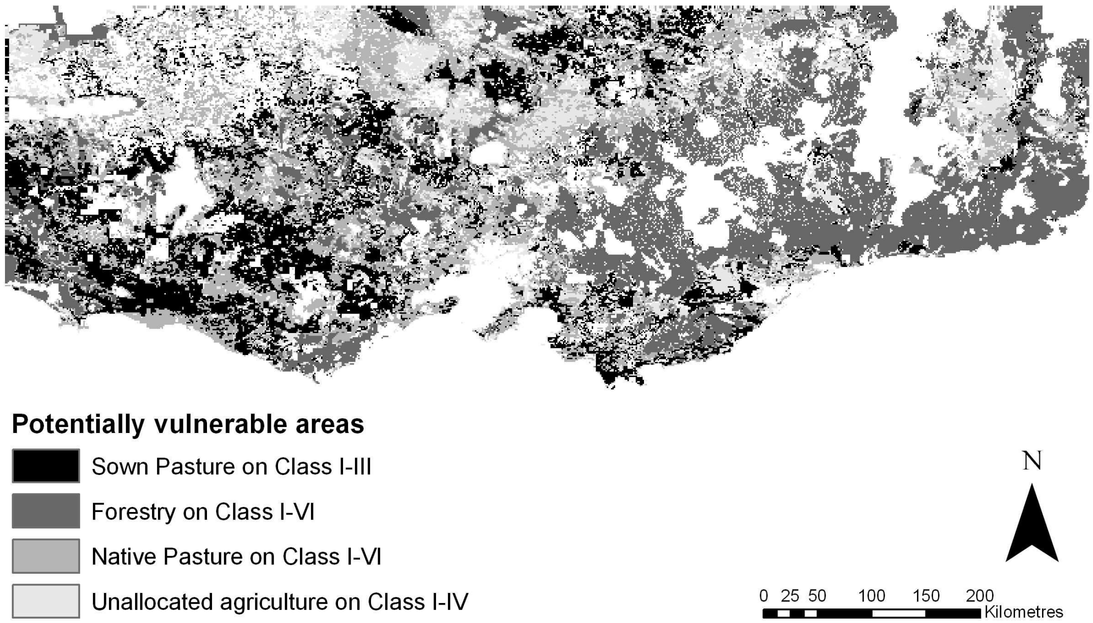


Figure 6.6. Potentially vulnerable land in the study area. Different shades indicate different sources of vulnerability as calculated from Table 2

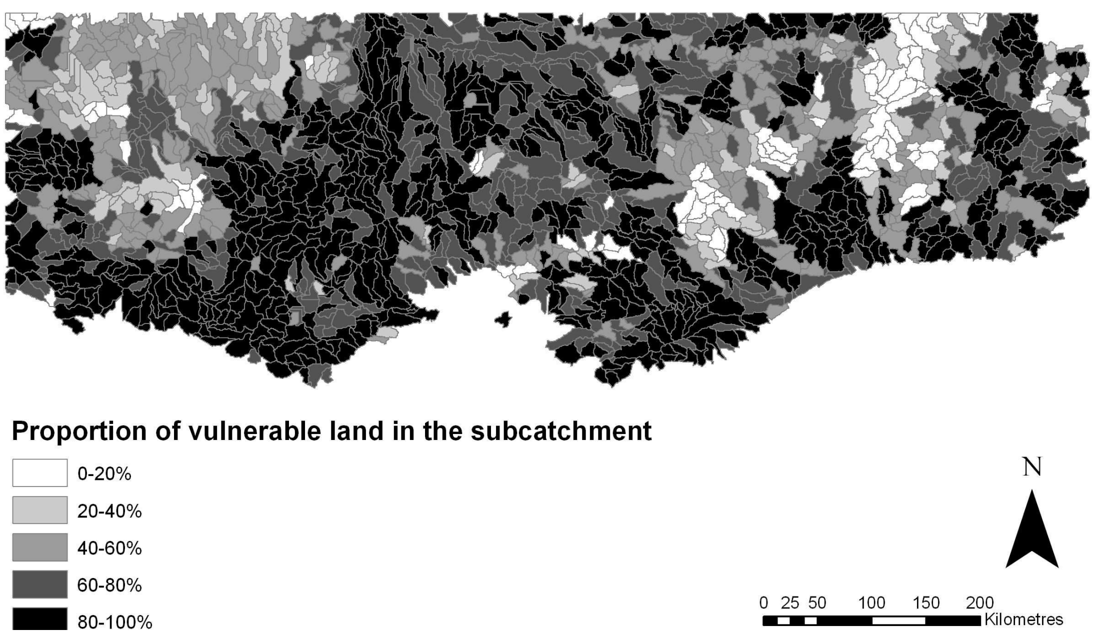


Figure 6.7. Vulnerability aggregated to catchments

6.4.4. Management integration

Although the SGA approach should be used as a continuous measure, we set a cut-off to categorise catchments in ‘good’ and ‘degraded’ condition. This cut-off was the 25th percentile (-0.012 on PC1, see dividing line in Figure 6.8), as two other reports - with independent methods – found that roughly 75-80 percent of river systems in Victoria are moderately to highly degraded (Ladson *et al.*, 1999; Norris *et al.*, 2001).

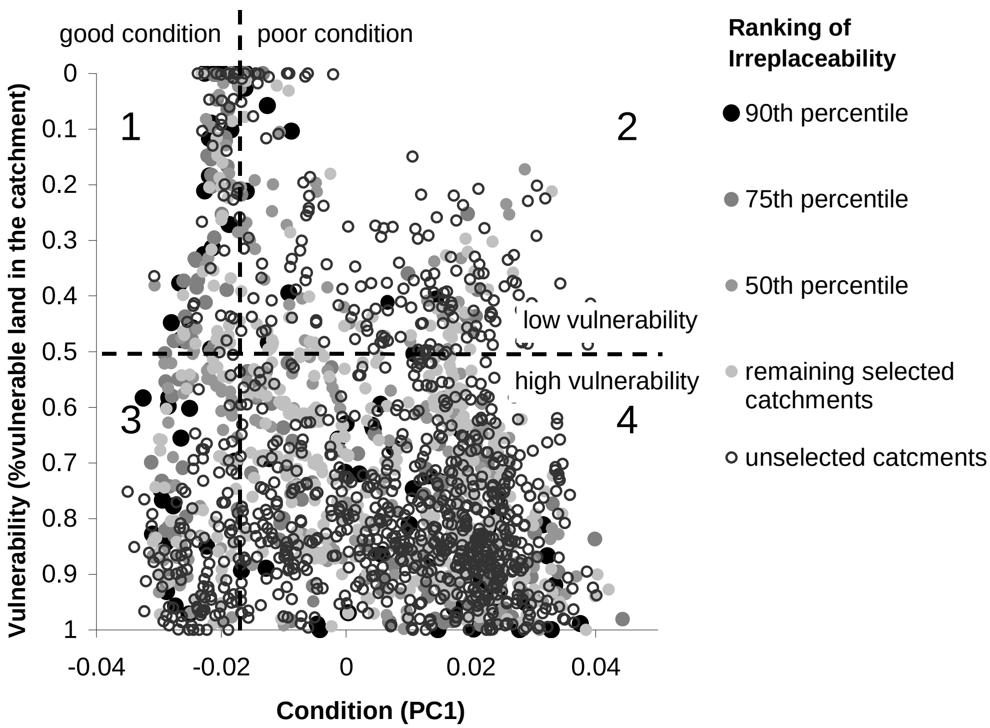


Figure 6.8. Plot of condition against vulnerability for the 1854 catchments. Darker dots indicate higher irreplaceability. The quadrants are analogous to Table 6.3.

Condition was plotted against vulnerability to classify the 1854 subcatchments into management categories as described in the decision matrix above (Table 6.3). The proportion of high irreplaceability catchments was reasonably even across the four quadrants. Most subcatchments (62 percent) were allocated to quadrant 4 (degraded condition/high vulnerability). These catchments – many located in the western part of the state - were mainly pastured land that would be suitable for cropping. These require restoration, together with the 11.5 percent catchments that are in degraded condition and have low vulnerability (quadrant 2). Catchments with degraded condition and high irreplaceability (as defined by the 75th percentile of selected catchments) are shown in Figure 6.9.

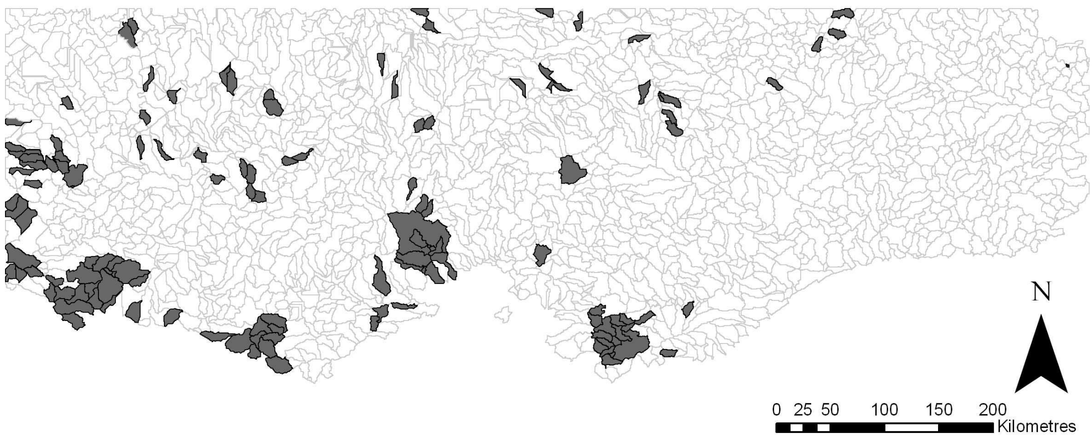


Figure 6.9. High potential irreplaceability subcatchments in degraded condition: priority catchments for restoration.

9.8 percent of the catchments were assessed as in good condition with low vulnerability. Most of these catchments are entirely on protected land managed for conservation. The remaining 16.8 percent of catchments in good condition but with high vulnerability are, however, on unprotected land, mostly low intensity grazing on native grasslands and mixed use forests. These catchments - especially ones with high irreplaceability (Figure 6.10) – are priority areas for assigning river reserves.

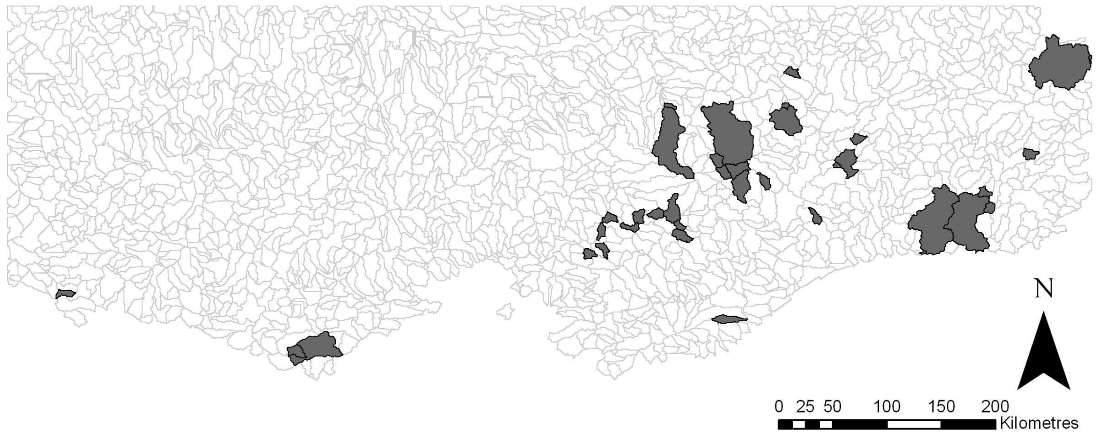


Figure 6.10. Subcatchments with high irreplaceability, good condition, and high vulnerability: candidate areas for river reserves.

6.5. Discussion

This study demonstrates a new, data-driven and repeatable way to approach three key questions in management of aquatic biodiversity. We consider the ICV (irreplaceability – condition - vulnerability) framework as an appropriate tool to meet the challenges outlined by Cullen (2003) and Dunn (2003). However – in line with Sarkar *et al.* (2002) - we emphasise that this is an academic study and no directions in conservation policy should be inferred for the state of Victoria. The main reasons for this are limitations in the indicator surrogates for irreplaceability – which have not been subjected to a proper target setting process (see Pressey et al., 2003) - as well as the simplified approach of assessing land capability and vulnerability which will be discussed below. Given proper interaction with managers and policy makers regarding conservation targets and appropriate consultation with land planners about vulnerability, the IVC framework will certainly be applicable for efficient real-world planning.

6.5.1. Irreplaceability

This study is one of the few attempts to model a large number of invertebrate taxa from GIS data only. Most predictive biomonitoring programs (Turak *et al.*, 1999; Clarke *et al.*, 2003; Ostermiller & Hawkins, 2004) use on-ground data such as temperature, basic chemistry and channel morphology along with remotely collected data. The main difference between our modelling approach and previous ones was our inclusion of variables that would be hard to acquire for the end-user of a bioassessment tool. These variables include topographic measures from IDRISI (Clark Labs, 2004) and estimators of soil and vegetation in the catchment. Many managers or consultants would not have access to these data layers or programs. Considering that five out of the six variables we used for modelling fall into this category, it is unlikely that we would have been able to model predicted distributions for 452 taxa without this extra information.

One weakness of our modelling approach was the omission of a validation dataset, used by Yuan (2004a) to avoid overfitting. This might also explain slightly inflated AUC values. However, we do not consider these issues as problematic for the purposes of our study. Discussed in detail by Linke *et al.* (in prep.), it is not the goal of the modelling exercise to accurately predict the presence of individual taxa in a catchment, but to model a general community structure that allows us to estimate its irreplaceability.

By calculating potential irreplaceability based on reference condition, we keep the irreplaceability component independent of condition and are able to flag catchments in need of restoration. An alternative approach would have been to modify the expected communities based on their actual condition. This was rejected on two grounds: first, it makes the irreplaceability assessment less transparent. If a taxon is not present, we would not know whether it was not expected in the first place or missing because of degradation. The second reason for dropping an integrated irreplaceability/condition metric was a philosophical one.

We wanted to avoid a situation similar to the Wild Rivers approach (Stein et al., 2002) where catchments in the best condition are by default the most irreplaceable.

Highly irreplaceable areas can be found in all environments, which indicates that the complementarity algorithm works well in achieving complete representation of targeted taxa. The highest concentration of highly irreplaceable subcatchments was in the sub-alpine region in the centre of the map (Figure 6.3). The temperate rainforests of the south-west and the adjacent agricultural regions were another irreplaceability hotspot (Figure 6.3). This illustrates a slight bias towards headwater and coastal catchments, introduced by the catchment configuration that did not allow non-headwater subcatchments to be selected alone. Although this bias is justified – it is a fact that most biodiversity can be conserved with least cost (area) in lower order catchments – we should include mechanisms that allow protection of mid- and lower order catchments in the future. As Pringle (2002) points out, not only upstream disturbance can affect aquatic biodiversity. Barriers for fish migration in middle and lower reaches represent an example of other relevant disturbances.

Target setting is one of the main challenges for an application of the ICV framework in planning and prioritisation. At this stage only invertebrates are used to estimate irreplaceability. Although invertebrates can be appropriate surrogates in biodiversity assessment (Heino et al., 2002), they operate on different spatial scales than other organisms that may have a higher commercial and social value (e.g. fish). Future applications of this framework will apply more thoroughly considered targets to a wider variety of organisms, which could also include assigning higher values to certain groups, in consultation with managers and policy makers.

6.5.2. Condition

The stressor gradient approach using pre-existing GIS layers proved a cost-effective and quick solution to a large-scale assessment. On a theoretical basis, the notion to let the data ‘speak for itself’ is both an advantage and a disadvantage. On first glance, the absence of a hierarchical framework (*sensu* Norris *et al.* 2001) or a rules-based approach similar to Walker *et al.* (2006) is a disadvantage. However, if Principal Components Analysis is used to summarize GIS layers, it has the potential to reveal hidden disturbance gradients that would have been otherwise masked by our pre-conceived conceptual model of what affects key characteristics of lotic ecosystems. PCA will also highlight cumulative effects and, by constructing orthogonal axes, select for stressors that work independently.

Most of the variation in the stressor variables could be explained by one axis (Table 6.4). Characterised by both cultivation and grazing as major land use categories, this axis can be summarised as an agricultural stressor gradient. Association of cropping and sown pasture with increased sediment budgets - described by Lenat & Crawford (1994) and Boulton & Brock (1999) – is confirmed by the high loading on this axis (Table 6.4). The second stressor gradient was characterised by urbanisation, reflected in the land use category and the high loading of both road density and nutrient levels on the PC axis. The link between urbanisation and nutrient levels in this particular area has been described in detail by Walsh *et al.* (2001). Although explaining 14 percent of variation, this very steep gradient is confined to the small area around Melbourne. The third gradient, describing smaller urban settlements and forestry has a relatively wide spread in the eastern part of the study region. We decided to omit both gradients from further analysis because of the small extent of the urban gradient and the relatively small impact that non-clearfelling forestry has on streams when compared to intensive agriculture (Lenat & Crawford, 1994; Norris *et al.*, 2001). However, in future assessments, an integration of different stressors should be developed to include all sources of disturbance into the ICV framework.

Although a stressor gradient approach does not automatically estimate ecosystem health, we are confident that in this study the principal components are adequately representing condition. Links between stressor layers – both land use categories and proximate stressors - and loss of taxa in benthic macroinvertebrate communities have been well established in both an Australian (Boulton & Brock, 1999; Gourley & Ridley, 2005; Norris *et al.*, in press) and international (Lenat & Crawford, 1994; Allan *et al.*, 1997; Allan, 2004) context. Furthermore, we found a significant correlation between the principal stressor and the Assessment of River Condition (Norris *et al.*, 2001; Norris *et al.*, in press). Some level of disagreement between the methods, which are based on completely different philosophies and data requirements, can be anticipated on the subcatchment scale, reflected in the relatively low r^2. This is analogous to the comparison of ARC_E with the Victorian Index of Stream Condition (Norris et al., in press). At the landscape scale, all three assessments agree strongly, as demonstrated in Figure 6.4. We also found a significant correlation between both the agricultural and the urban stressor and AUSRIVAS (Simpson & Norris, 2000), a RIVPACS-like method to assess loss of common benthic taxa. The AUSRIVAS dataset was strongly biased to catchments in pristine condition, which possibly lowered the r^2 of the correlation (0.18). While we recommend an unbiased sampling design to ground-truth the link between stressor gradients and loss of taxa, we are confident that condition is adequately represented in this study.

Although it is one of the main strengths of the SGA to describe stressors as gradients, we used the correlations between ARC, ISC and our method to set a threshold. With both the ISC and ARC_E estimating the proportion of degraded streams at 75-80 percent, we chose a conservative 25th percentile as our cut-off for the decision matrix. If used in a planning scenario, this would be flexible and based on expert opinion, considering the taxa or other features targeted for conservation action.

6.5.3. Vulnerability

The inclusion of vulnerability – recognised in terrestrial planning for over a decade – is a new concept for aquatic ecosystems. We used a combination of traditional methods to estimate exposure from land capability (Pressey & Taffs, 2001; Rouget *et al.*, 2003a) and a conceptual model ranking the influence of different land uses on Australian streams (Norris et al., 2001).

To keep this study as simple as possible, land capability was only estimated from three key parameters: slopes, erosion risk, and salinity risk. Although it is consistent with methods described in Emery (1985) and Pressey & Taffs (2001), this might have led to slightly overestimating the extent of land capable for cultivation, because additional factors will control the agronomic feasibility of cropping. A future assessment could include soil limitations and water availability, which could reduce vulnerability ratings in the south-western region of the study area (Figure 6.6). A second source of bias in our approach of estimating likely exposure was the tenure of forestry. Most of the forested areas we considered as vulnerable to conversion to pasture or cultivation are actually unreserved public land, including many state forests. The likelihood of these publicly tenured catchments being cleared is considerably lower than for comparable private land. Accounting for this in future work will lower vulnerability values in subcatchments with this tenure, although the relationship between land capability and potential intensification of logging will also need to be considered.

To estimate vulnerability more accurately in future assessments, a measure of intensity as defined by Wilson *et al.* (2005a) would be useful. Using the framework in Table 6.1, the introduction of low intensity forestry to a natural area will cause less degradation of stream condition than clear-cutting and replacing the forest with cropping. Another scenario would be conversion of forest to native pasture, which will have less effect than conversion to tilled and irrigated land.

Mapping vulnerability revealed an interesting pattern: most subcatchments contain more than 50 percent vulnerable land. Two exceptions can be found in the central-east and the northwest of the study region, yet for completely different reasons. Many catchments in the central-east are in protected areas or on low-capability land. These often go hand-in hand. Historically in Australia, as elsewhere, much of the land deemed unsuitable for agriculture and inaccessible for forestry operations was declared as a protected or 'wilderness' (Pressey et al., 2000). We also assigned a low vulnerability rating to the catchments in the northwest (Figure 6.6), most of which have a high proportion of irrigated cropping or other high intensity agriculture. Neglecting management practices, in our framework (Table 6.2) these are classified into the highest land-use category, so no further degradation is possible.

6.5.4. Management integration and conclusions

Using condition and vulnerability to prescribe different courses of action and irreplaceability to set spatial priorities, the ICV framework integrates concepts from stream bioassessment and systematic conservation planning in terrestrial environments. While the aquatic field has focussed on the assessment of river condition, vulnerability has been assessed in a terrestrial context for over a decade (Wilson et al., 2005a). While other systems for river conservation management integrate multiple dimensions into a single index value (SERCON, Boon et al., 1998), we believe that the ICV framework can be used differentiate between required conservation actions.

Recently, several authors (Cullen, 2003; Dunn, 2003; Fitzsimons & Robertson, 2005; Kingsford *et al.*, 2005) have renewed calls to establish a comprehensive, adequate and representative (CAR) reserve system for Australian rivers. The ICV framework assists with the application of the criteria for CAR reserves. Depending on the data used for the complementarity-driven algorithm, irreplaceability values contribute to comprehensiveness (the coverage of river types by protected areas) and representativeness (the biological variation within river types, indicated by taxa or approximated by the occurrences of river types in different regions or subregions (NRMC, 2004). Adequacy is addressed by the whole-of-catchment approach to protection, in which non-headwater subcatchments by themselves are not allowed configurations. Extended data sets and refined conservation targets would improve the application of all three criteria. However, by including an assessment of vulnerability, the ICV framework goes beyond CAR by indicating the relative urgency for protection or restoration. The British SERCON (Boon *et al.*, 1998; Boon *et al.*, 2002) already integrates irreplaceability and condition, but like the Wild Rivers approach in Australia (Stein et al., 2002) or bioregionalisation strategies (Fitzsimons & Robertson, 2005; Kingsford *et al.*, 2005) it neglects vulnerability. Catchments with low vulnerability – by definition – have a small likelihood of further degradation in the near future. While monitoring is essential, the main conservation effort (i.e. reserve design) should be directed to areas that are highly susceptible to future degradation (Margules & Pressey, 2000). In our study area, these potential river reserves are highlighted in Figure 6.10.

In degraded catchments – 80 percent of our study area - management options are not straightforward. In this initial version of the ICV framework, we assume that restoration will return a modified river back to its natural state. However, the outcome will most likely not be the “original” ecosystem, depending on the historical baseline agreed by managers. The difficulty of reinstating a previous condition has been discussed extensively in theory (Hobbs & Norton, 1996) and demonstrated by the restoration of the Rhine, which is now home to dozens of invasive taxa (Tittizer *et al.*, 2000; Freyhof, 2002). Therefore, we might need to modify the concept of ‘potential irreplaceability’ modelled from undisturbed catchments as an estimator of conservation value in the ICV framework. Modelling that considers the type and intensity of disturbance could produce an expected community composition after restoration or at least flag taxa and other features that are unlikely to be present after rehabilitation. This information could be used in the derivation of a modified irreplaceability index. In the meantime, we believe that prioritising restoration in the ICV framework, despite ignoring hysteresis effects in re-colonization, is a large improvement over educated guesses.

In this application of the framework, the irreplaceability component is independent of the other two axes, visualised in Figure 6.8, where the proportion of highly irreplaceable subcatchments is roughly the same in every quadrant. Together with the even spread of highly irreplaceable catchments throughout the landscape, this confirms that irreplaceability is a completely independent measure, both spatially and in relation to condition and vulnerability. In contrast, condition and vulnerability are not independent from each other. Land use, related to condition, is used to derive vulnerability. This leads to some land use values defaulting to lower vulnerability ratings. In our view, this lack of independence does not compromise the assessment, but reflects the realities of present and potential uses and their effects on rivers.

The ICV framework should be recognized as just that – a framework to promote structured discussion and decisions, instead of a recipe for uncritical adoption. One of its major strengths is its flexibility. Assessments of irreplaceability can be based on many data sources, apart from the modelled distributions of taxa. Existing assessments of conservation value, such as bioregionalisations (Fitzsimons & Robertson, 2005) in data-poor areas can be used as the features of conservation interest. Analogous to this, instead of the stressor gradient approach, other assessments of ecological condition can be used. Previous assessments in Australia include the Assessment of River Condition (Norris *et al.*, 2001; Norris *et al.*, in press) and the Wild Rivers project (Stein et al., 2002). International large-scale assessments that would be suitable include EMAP (Hughes et al., 2000), SERCON (Boon et al., 2002) and the forthcoming Assessment of Wadeable Streams in the USA (USEPA, 2004).

Still a young field, planning for river conservation has been increasing its profile over the last five years, with prominent statements by scientists and conservation groups (Abell, 2002; Cullen, 2003; Dunn, 2003; Kingsford *et al.*, 2005). The ICV framework is a data-driven and repeatable (Duelli, 1997) approach to tackling the unresolved issues of prioritising conservation and restoration efforts. Founded in good science, it should help to convey decisions about priority areas to managers and the public. Along with other initiatives developing explicit approaches to river conservation (Boon *et al.*, 2002; Fitzsimons & Robertson, 2005), we hope to contribute to fulfil Brian Moss's (1999) wish regarding the conservation of river biodiversity: 'The sixth age has been fool's gold; the seventh age could be the real stuff.'

Chapter 7. Synopsis

7.1. Introduction

It was the aim of this thesis to address existing gaps in freshwater conservation planning (as outlined in Chapter 2) and to introduce modern techniques used in systematic conservation planning for terrestrial systems to a riverine context. Specific goals and properties of a new framework were:

1. An estimator of conservation value that is either based on complementarity or corrects for site-specific rarity
2. A biota-driven estimator of conservation value, with flexibility to include abiotic surrogates or ecological processes
3. A comprehensive framework including vulnerability and a new independent axis to measure condition as a descriptor for the nature of conservation action
4. Acknowledgement of the connected nature of rivers and the need for whole-catchment protection.

7.2. O/E BIODIV

The first approach - O/E(BIODIV) - fulfils goal 1 and partially fulfils goals 2 and 3. O/E(BIODIV) is not complementarity-based, but uses a site-specific measure of rarity, driven by both global occurrences (in the study region) and the predicted taxa at a site. This is an improvement over simple measures of richness or rarity that usually favour one habitat over another. In most types of ecosystems, there are taxa-rich and taxa-depauperate areas – in rivers, for example, most mid-order streams are richer than headwater or lowland streams (Minshall et al., 1985). To correct for bias towards naturally rich areas, a rare taxon in the context of O/E(BIODIV) is a taxon not expected at a site (i.e. a low probability of occurrence). The actual metric is a stochastic measure of how many rare taxa are expected at a site compared to how many are observed. Therefore, a site is assessed in the light of its ecological potential, rather than an arbitrary standard.

The two-tiered framework outlined in Chapter 3 serves as a blueprint for the integration of condition assessment with the assessment of conservation value. This prescribes a direction for the action required (protection or restoration) and maximises transparency. Are both common and rare taxa richness naturally low or has an impact caused a decline? Apart from providing an answer to this question, the two-tiered approach removes the *a priori* bias towards undisturbed areas, described by Mace *et al.* (2000). By integrating condition and conservation assessments, even disturbed areas can be assessed as potential hotspots.

Still, there are major limitations associated with the approach of O/E(BIODIV). Although the two-tiered framework includes a measure of condition, an estimate of vulnerability (i.e. how prone is the site to future degradation) is missing from the framework. Also, the measures of both condition and conservation value in the two-tiered framework are based on site-measurements. Despite reasonable coverage of both invertebrate and fish distributions in existing North American and Australian databases (Hughes *et al.*, 2000; Simpson & Norris, 2000), the site-based nature of the approach makes extrapolation and large-scale planning efforts difficult. It will be by definition impossible to model species defined as ‘rare’ by O/E(BIODIV) across the landscape – only taxa that cannot be modelled with sufficient certainty enter the index. However, spatial extrapolation methods could be used in future studies to estimate final values of O/E(BIODIV) to assess unsampled areas.

The biggest strength of O/E(BIODIV) is also its biggest weakness: the site-specific definition of ‘rare’ taxa. For the above definition to hold up, the underlying RIVPACS or ANNA model will need to explain a large amount of the natural variation (Chapter 4). ‘Rare’ taxa (as defined by $p<0.5$ likelihood of occurrence) will really have to be ‘unexpected’ or not able to be modelled. If taxa are not predicted because of model error or artefacts of the RIVPACS grouping step, the concept of O/E(BIODIV) will collapse, common taxa will be declared as rare and will hence change the metric. Furthermore, if not many taxa are predicted, O/E(BIODIV) will turn into a plain richness index .Therefore – despite an attractive theoretical grounding, the application of O/E(BIODIV) will be restricted to datasets where strong environmental gradients explain a very large amount of variation in the data (Chapter 4), which will in most cases be difficult to achieve.

7.3. Irreplaceability

An approach based solely on complementarity does not implicitly correct for biases in richness and rarity such as the richness gradient between lower- and mid-order streams noted above. However, areas with naturally low richness will not be neglected if they provide maximum complementarity. Furthermore, the selection algorithms will not stop before all taxa (or other targets) are selected.

Unfortunately, very rare taxa are not considered in the complementarity algorithm described in Chapters 5 and 6. The reason for this is the modelling step with which taxa distributions are projected across the landscape. To avoid extreme cases of overfitting, taxa with fewer than 10 observations across the dataset were omitted before building the generalised additive models with which taxa distributions were estimated (Chapter 6). As databases grow with extensive assessment programs being adopted by many jurisdictions, more taxa could potentially be modelled in the future using this approach. Also, with more extensive data, validation datasets could be withheld to estimate the degree of overfitting and its influence on reserve selection procedures.

Two factors highlight the need to develop a systematic planning approach especially geared to rivers instead of simply adopting a terrestrial approach. First, the maps comparing the minimum set with- and without catchment protection illustrate the errors that a simple adoption of the grid-cell method common in terrestrial planning would cause. Second, the mathematical properties of the heuristic algorithm change with the spatial configuration of the upstream catchment protection. The classic MNP ‘progressive rarity’ algorithm (Margules et al., 1988) is no longer meaningful when area is used as a cost factor. A blind adoption of this algorithm – the most efficient in terrestrial settings (Csuti et al., 1997) – would have caused a 100 000 hectare error in the best case of the minimum set (Chapter 6).

The modified algorithm used here (Chapter 6) also led to the development of a new estimator of irreplaceability (Chapter 6). The bootstrapped estimator offers both a transparent representation of irreplaceability, and an insight into multiple facets by distinguishing between different types of catchment: The average contribution to conservation targets - in this study, the protected taxa - distinguishes catchments with maximum conservation value from those with diminishing complementarity. However, even a catchment with few contributing taxa can be selected in every bootstrapped run if it has a unique combination or endemic taxa and therefore scores high irreplaceability values.

A taxon or feature-based algorithm is superior to an ecoregions approach whether complete biodiversity protection or prioritisation of hotspots is the aim of a planning exercise. As a justification to their ecoregion-based approach, Higgins *et al.* (2005) claim that methods using taxa distributions are biased towards areas with very rare or endemic taxa. An examination of the different classes of catchment irreplaceability (Chapter 6) reveals that this is not the case in the approach adopted here. A catchment with only a few very rare or endemic taxa would be chosen in every solution, but contribute few taxa. It would therefore be in the list of irreplaceable catchments, but not in the highest percentile of contributions – hence at lower priority than catchments with high complementarity. While a taxon, or features-based, approach is imperative to a conservation plan in which all targets have to be met, even a coarse prioritisation exercise – as proposed by Higgins *et al.* (2005) - will work using the proposed algorithm (Chapter 6).

7.4. Systematic conservation planning in riverine landscapes

With the addition of condition and vulnerability, this study emerged from the initially planned assessment of river biodiversity to a comprehensive framework for river conservation planning (Chapter 6). After identifying areas of high conservation value, the nature of conservation measures – protection or rehabilitation - can be determined by the condition axis of the assessment (Chapter 6). Identifying the most vulnerable of these catchments then ensures maximum efficiency for conservation action.

In contrast to the condition stage of the site-scale two-tiered framework around O/E(BIODIV), the condition axis of the ICV framework developed here is geared towards landscape-scale assessment. The stressor-gradient approach appears capable of rapid and low-cost large-scale assessment that corroborates well with established methods such as ARC (Norris et al., in press) and ISC (Ladson et al., 1999). In a time where many natural resource management agencies have their own trialed and tested approaches, the ICV framework leaves enough flexibility to incorporate existing large-scale assessments. Existing assessments that could be used include ARC (Norris et al., in press) and ISC (Ladson et al., 1999) in Australia, EMAP (Hughes et al., 2000) in the USA or the condition components of SERCON (Boon *et al.*, 1998; Boon *et al.*, 2002).

Vulnerability to land change has been identified as a key component in previous applications of the terrestrial I-V framework (Pressey & Taffs, 2001; Cowling *et al.*, 2003a). Nevertheless, application of the vulnerability component in a riverine context will need to include other types of threat, such as invasive species or increased intensity of present exposure. Again, the flexibility of the ICV-framework enables existing assessments of vulnerability to be incorporated into a conservation plan. However, any assessment of vulnerability should be in line with existing conservation frameworks (Wilson et al., 2005a). These frameworks state clearly that a threat to biodiversity will imply a temporal component of future degradation. Water specific challenges include estimates of abstraction and regulation, as well as future changes in connectivity. Although these challenges have been included in the analysis that led to impact classifications of different land use classes (Norris et al., 2001), a more explicit formulation would be desirable. This would include a distinction between proximate threats and long-term effects and include scale-dependencies between different sources of vulnerability.

Some existing assessments of vulnerability –like the US EPA's Regional Vulnerability Assessment Program (Smith et al., 2000)– also consider present condition, which will cause conflict with the condition axis in the ICV framework. However, components of this program could be used for condition and vulnerability respectively.

7.5. Directions for future research

The Yukon dataset (Chapter 4) highlights that, despite the flexibility of the methods, not all datasets are usable with the approaches presented here. The Yukon study is characterised by low taxa richness at the site level, but high β-diversity across the dataset. With little structure in the data, confidence in O/E(BIODIV) is low, because many 'rare' taxa could just be common taxa that are not predicted because of poor model performance. On the other hand, a measure of irreplaceability at a landscape scale could not be calculated either because only 27 reasonably common taxa had more than 10 observations across the dataset (Chapter 6). Therefore taxa could not be extrapolated to other catchments using GAMs, which is necessary to build the input matrix of the complementarity algorithm. While Roux et al. (2002) use environmental surrogates – a widespread technique in terrestrial studies(Lombard *et al.*, 2003; Sarkar *et al.*, 2005) – others have suggested that environmental surrogates are often insufficient when trying to represent biodiversity in conservation planning (Araujo et al., 2001). Even highly stratified species-habitat models are often not sufficient to ensure adequate representation by environmental descriptors (Altmoos & Henle, 2007). Environmental diversity (ED, Faith, 2003; Faith *et al.*, 2004) and generalised dissimilarity modelling (GDM, Ferrier & Guisan, 2006) represents a potential solution to be explored in future projects. Hereby multivariate environmental similarity is linked directly to species patterns to create environmental surrogates that can be used for prioritisation.

Setting conservation targets to maximise persistence is another vast area for future research. While the ICV framework provides the flexibility for rule- and target setting, these rules and targets will have to be populated by researchers and managers to ensure that the main vehicle to biodiversity conservation is met: persistence at the population level (Gaston et al., 2002). Conservation of fish populations, for example, needs to ensure that key habitat requirements -- including migration and spawning - are met (Abell, 2002) and connectivity is maintained. Setting correct targets will also ensure that issues of scale - arising from protecting multiple taxonomic groups – are adequately taken care of. It is obvious that scales of influence on fish conservation are different to macroinvertebrate and plant conservation. A proper target setting procedure will implicitly take care of this issue.

While the a whole-catchment approach in the ICV framework is a definite improvement over existing methods for river conservation planning, it will have to be critically evaluated in future research. The strict rules for inclusion of the entire catchment upstream are impractical in the case of large lowland rivers. Some disturbance types might not even need full catchment protection – organic pollution for example (Storey & Cowley, 1997). A modification of the catchment rule depending on the stressor and incorporation of mixed models of protection (Saunders *et al.*, 2002; Cullen, 2003) would make the framework more feasible for large rivers. Furthermore, evaluation of cross-catchment (Pringle, 2001) and downstream influences (Pringle, 2001; Yates & Bailey, in press-a) will potentially introduce additional modifications to the way river conservation interfaces with holistic catchment planning.

Acknowledging both – multiple target groups and multiple modes of conservation action – will be an integral part of a properly working systematic conservation framework. In such a framework, specific threats would act on specific targets. An example would be that some targets would be satisfied by selecting a smaller area and prescribing a lower lever of protection – mixed use or stewardship, while other organisms might require strict reservation of larger areas.

7.6. Conclusion

Between the two approaches (O/E BIODIV and IVC) developed in my thesis, all four goals and properties stated at the beginning of this chapter were met. O/E(BIODIV) dealt better with rare taxa – given that RIVPACS or ANNA models explain a large amount of variation - while the ICV framework using a complementarity based estimator was flexible towards target setting (goals 1 and 2). The ICV framework also drives measures and priorities at a landscape scale (goal 3) and considers catchment-scale effects (goal 4). An integration of O/E(BIODIV) into the ICV framework would be ideal. The flexibility of the ICV framework means that it could be integrated with O/E(BIODIV) provided good models and a landscape extrapolation of O/E(BIODIV). With this flexibility to integrate with existing components and the catchment-scale orientation, the ICV framework finally brings freshwater conservation planning up to speed with terrestrial approaches and has the potential for future worldwide use.

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