

**A continental landscape framework for
systematic conservation planning for
Australian rivers and streams**

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September, 2006

**A thesis submitted for the degree of Doctor of
Philosophy of the Australian National University**

I certify that unless otherwise acknowledged this thesis is my own original work. It includes material also published in the Proceedings of the Fourth Australian Stream Management Conference (Stein, 2005) and a report to Land and Water Australia (Kingsford *et al.*, 2005).

Acknowledgements

The assistance and support of many people is very gratefully acknowledged. In particular, I thank my principal supervisors, Professors Michael Hutchinson and Henry Nix for their support and encouragement over many years and insightful comments on the thesis draft. The other two members of my supervisory panel, Professor Bob Wasson and Dr. Jon Olley from CSIRO Land and Water, also gave helpful advice. CSIRO Land and Water is thanked too for financial support in the form of a top up scholarship to supplement an Australian Postgraduate Award.

This research was inspired by concern for the future of non-humans in a world increasingly dominated and shaped by humans. Its approach is strongly influenced by my early research experience, first at CSIRO Land Use Research (now Land and Water) with Mike Austin, Chris Margules and Dan Faith and then at CRES under Henry Nix and Mike Hutchinson. I am also grateful to the members of the working group involved in the preparation of a report for Land and Water Australia on the protection of high-conservation value rivers: Richard Kingsford, Jon Nevill, Helen Dunn, Jim Tait, Peter Davies, Jane Doolan, Debbie Love and Sarah Fairfull. The enthusiastic debates within the working group encouraged me to clarify and critically review my ideas and broadened my perspective. I also thank Stuart Blanch (WWF) for helpful discussions and encouragement and Jon Nevill (Only One Planet Consulting) for his dedication to the cause of river conservation and suggestions for the presentation of the analysis of the NRS.

Many others also provided advice. Ian White (CRES) and Barry Croke (ICAM) assisted with suggestions on hydrological modelling while Sara Beavis (CRES) and Colin Pain (CRCLEME) made useful suggestions on the inclusion of geological attributes. John Gallant (CSIRO Land and Water) generously made available his program for computing the index of valley bottom flatness and provided advice on its use. Advice on statistical methods and presentation was provided by members of the ANU Statistical Consulting Unit (Jeff Wood, Emma Knight and John Maindonald) and Ross Cunningham (CRES).

I am very grateful to John Stein and Fred and Ida Argy who shared the task of proof-reading. This research relied on data developed at CRES and elsewhere. I especially thank Michael Hutchinson and John Stein for access to an interim revision of the national 9 second DEM version 2 and streamlines derived from the 5m Cotter ALS DEM. The fish survey data used to test the ecological relevance of the river environment classifications were supplied by Simon Hartley from NSW Fisheries. Simon also willingly answered my many queries. Simon Townsend (Northern Territory Department of Infrastructure, Planning & Environment) and Jane Hosking from the Commonwealth Department of Environment and Heritage supplied the National River Health data. Robert Smart (Bureau of Rural Science) organized access to the

Integrated Vegetation dataset. Albert Derigo (ACT Land Information Centre) provided the streamline mapping from the ACT 1:10,000 scale topographic map series.

CRES provided a friendly and stimulating environment for study. I received support from fellow students and staff and new ideas and understanding from a rich diversity of disciplines. I would like to thank the CRES support staff for assistance with finance, travel and other administrative tasks. Mark Greenaway and Phil Greaves are especially thanked for ready advice on computing problems and most importantly, ensuring the operation of the CRES computing system.

Finally and most importantly I thank my family: my children, Patrick and Sarah who ensured some balance in my life; my parents, Ida and Fred who supplied regular newspaper clippings to keep me informed on river related news; my dogs, Jack and Jessie who made sure that if I did nothing else every day I went for a walk and above all, my husband and colleague, John. This thesis would not have been possible without their encouragement, support, and most of all, patience.

Abstract

Conservation of Australia's distinctive river ecosystems has lagged behind that of terrestrial and marine environments despite mounting evidence of the destructive effects of human activities. There has been little nationally coordinated conservation activity. A systematic, continent-wide conservation planning approach would ensure limited conservation resources are allocated efficiently and decisions are accountable. This thesis addresses critical gaps in the spatial data required for systematic conservation planning. It describes the development of a stream network and nested catchment reference system to provide the spatial framework. This framework, at a map scale of approximately 1:250,000, supplies planning units for application of reserve design algorithms and determination of priorities for protective management as well as units for reporting conservation evaluation and assessment. The Pfafstetter coding of the catchment units identifies drainage network connectivities allowing them to be readily incorporated into conservation planning procedures.

This thesis presents for the first time a comprehensive picture of continent-wide variation in the landscape factors that ultimately control riverine ecosystem patterns and processes. Stream segments, the section of the stream between tributary confluences and the smallest unit in the spatial framework, have been individually characterized and classified at multiple scales using attributes that describe the catchment (and/or sub-catchment) climate, water balance, geology, terrain and vegetation. Segments were clustered according to their similarity in environmental data space so that, unlike ecoregion classifications, groups may be geographically dispersed. The resulting River Environment Types have been found to differentiate significant variation in stream biota (macroinvertebrates and fish) and habitat characteristics. However, classification strength varies widely among types. It is hypothesised that this is due to both classification uncertainty and limitations of the test data.

The spatial framework and classifications are the central elements of a continental landscape framework that could be used to support systematic conservation planning and assist the development of a national conservation plan for Australia's rivers and streams. The framework could also provide the basis for an online information system to serve a broader range of NRM planning and management objectives. The utility of the continental framework has been demonstrated in a review of the National Reserve System (NRS). The review acknowledges some uncertainty in the results due to inaccuracies and limitations of the framework, but nevertheless, concludes that the NRS must be expanded if it is to achieve a comprehensive and adequate protected area system for river ecosystems.

This study makes major contributions to spatial analysis methodology. It has developed and applied a new method of drainage analysis applicable to diverse drainage structures at continental scale and recommends enhancements to the internationally adopted Pfafstetter scheme. It also advances understanding of the role for landscape classification and the influence of classification choice on conservation planning outcomes. The development of the continental landscape framework for Australia presents a model and the necessary tools for conservation planning for the rivers and streams of other continents.

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List of Abbreviations

ACT	Australian Capital Territory
AIC	Akaike Information Criterion
ALS	Airborne Laser Scanning
AML	Arc/Info Macro Language
ANRDL	Australian Natural Resources Data Library
ASRIS	Australian Soil Resources Information System
AusRivAS	Australian River Assessment Scheme
AWRC	Australian Water Resources Council
BIC	Bayesian Information Criterion
CAPAD	Collaborative Australian Protected Areas Database
CAR	Comprehensive, adequate and representative
CFEV	Conservation of Freshwater Ecosystem Values
CRES	Centre for Resource and Environmental Studies
CS	Classification Strength
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DEM	Digital Elevation Model
ECS	Evaluated Corridor Section
GIS	Geographic Information System
IBRA	Interim Biogeographical Regionalisation of Australia
IUCN	International Union for the Conservation of Nature and Natural Resources
mrRTF	multi-resolution Ridge Top Flatness Index
mrVBF	multi-resolution Valley Bottom Flatness Index
NHRP	National River Health Program
NLWRA	National Land and Water Resources Audit
NRM	Natural Resource Management
NRS	National Reserve System
NSW	New South Wales
NVIS	National Vegetation Information System
RDI	River Disturbance Index
RFA	Regional Forest Agreement
SA	South Australia
SERCON	System for Evaluating Rivers for Conservation
WA	Western Australia

Chapter 1. Introduction

Much is made of the general aridity of Australia. As the driest inhabited continent, Australia has the “lowest percentage of rainfall as run-off, the lowest amount of run-off, the least amount of water in rivers and the smallest area of permanent wetlands “(State of Environment Advisory Council, 1996, p.7-4). Nonetheless, examples of almost all types of freshwater environments are found in Australia (Boulton and Brock, 1999). Section 1.1 introduces the diversity of river forms and processes that may be found. Mary White’s beautifully illustrated book ‘Running Down: Water in a Changing World’ (White, 2000) offers a comprehensive account. She portrays the many human activities that threaten these rivers and presents compelling arguments for conservation action. This thesis addresses that concern. It describes the development of a continental landscape framework for Australian rivers to underpin a national program for the conservation of riverine ecosystems.

This chapter provides the background to this task. It highlights the distinctive characteristics of Australian streams that lend them global significance (Section 1.2). These distinguishing, often unique, features not only strengthen the case for conservation but also limit the choice of spatial analysis methods. Section 1.3 briefly describes the poor state of many rivers, and the increasing threats to others, to emphasise the immediacy of the problem. Although there have been conservation initiatives at both Commonwealth and State and Territory levels these are shown to be insufficient and poorly coordinated and implemented (Section 1.4). The case is therefore made for a national conservation plan with a planning approach that provides for a rational allocation of scarce conservation resources to maximise conservation benefits (Section 1.5). The chapter concludes with the specific aims of this research and the structure of the thesis.

1.1. Australian rivers and streams

Average annual runoff is very low in large parts of the continent due to low rainfall and very high rates of evaporation, yet there is great diversity among rivers of the arid zone. The streams of the Western Plateau region of Western Australia (Figure 1.1) are remnants of the much wetter Tertiary period and now are ephemeral and largely disconnected (Mabbutt and Sullivan, 1970). Broad valleys, now containing numerous salt lakes, follow ancient drainage lines that once originated in Antarctica, before continental drift separated the two continents (Hutchinson and Dowling, 1991; Ollier, 2001). Many of the desert rivers end in extensive dunefields. Their channels have been diverted by dune barriers and absorbed by the sands (Mabbutt, 1986). In the north of the Simpson Desert, for example, sand dunes divert the Hale and Todd Rivers along the swales for more than a hundred kilometres while in the south, the terminal floods of the Finke River backup between the dunes (Mabbutt, 1986). The “Channel Country” is so named for the

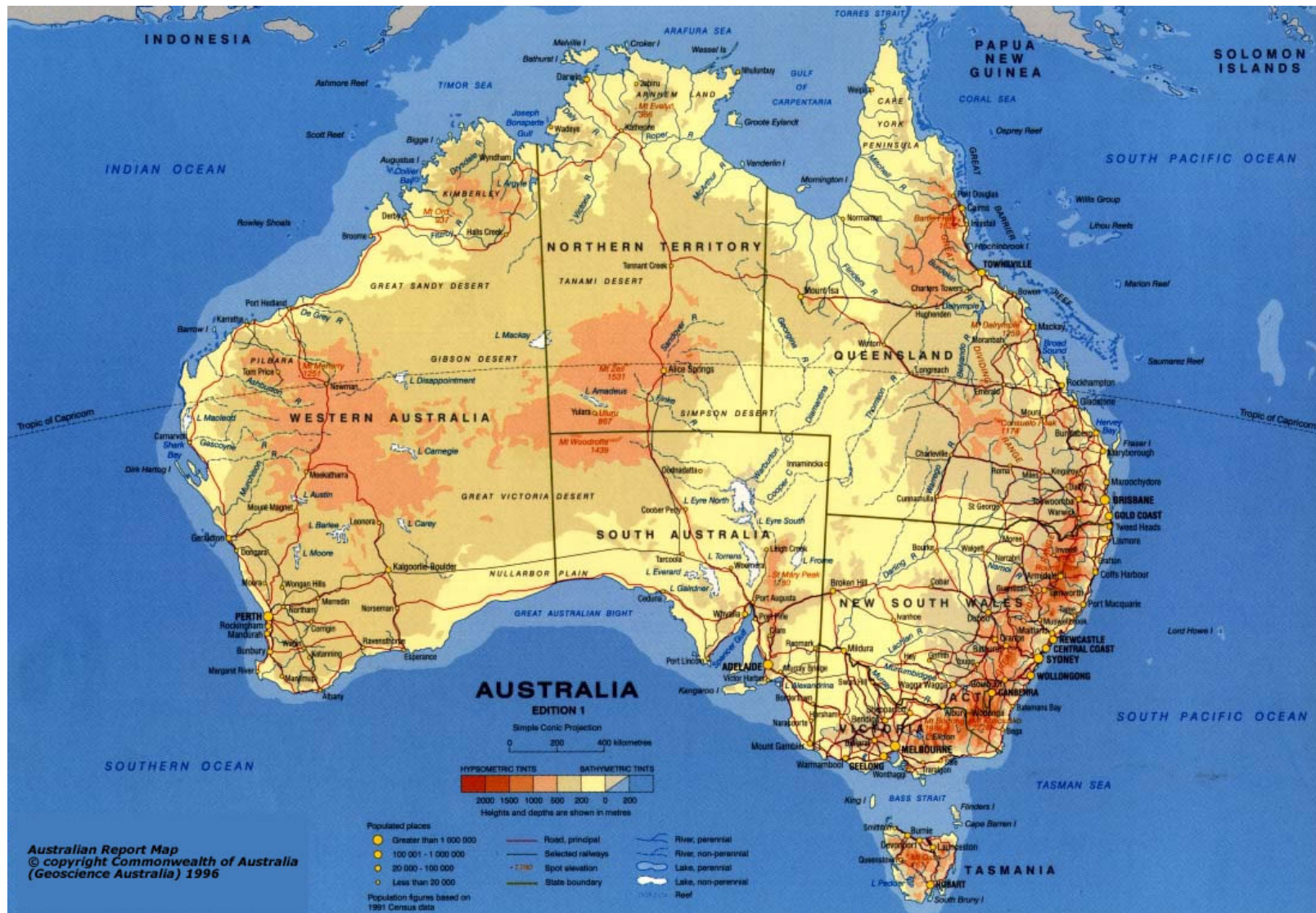


Figure 1.1. Locality map showing State and Territory borders and major rivers.

extensively anastomosing system of channels that characterize the floodplains of the Diamantina and Georgina Rivers and Cooper Creek in the Lake Eyre Basin. The channel belt occupies a floodplain that may be more than 50 kilometres wide (Costelloe *et al.*, 2003) with many permanent waterholes (Knighton and Nanson, 1994b). Extensive floodplains are also characteristic of the rivers of the Murray Darling Basin, Australia's largest coastal draining basin. The basin is comparable in area and river length with major river systems of the world but has a much lower discharge (State of Environment Advisory Council, 1996) as much of the system flows through arid areas that contribute little or no runoff (Young, Schiller, Roberts *et al.*, 2001). A number of tributaries of the major rivers terminate in a deltaic distributary channel system with only infrequent connection to the main river (Young, Schiller, Roberts *et al.*, 2001). The Paroo, for example, only reaches the Darling River in large floods (Young, 1999) and as a result, populations of many aquatic species may be relatively isolated from other parts of the Basin (Watts, 1999).

The rivers of northern Australia have a predictable seasonal variability in discharge associated with the tropical monsoon. Wet season peak flows can be very high and produce extensive flooding. On the vast floodplains, rivers divide into numerous anabranches and wetlands, including many listed on the Directory of Important Wetlands (Environment Australia, 2001), and support large numbers of birds and an abundance of aquatic life (Arthington *et al.*, 2004).

The summer dominance of flows in the north gradually shifts to winter/spring dominance in the south, most notably in the isolated rivers of the south-west (McMahon, 1979). The south-west rivers are relatively short, most flowing within 50 to 150km of the coast (Pen, 1999) and have a distinctive fauna (Bunn and Davies, 2000). Groups of invertebrates found in similar streams in south-eastern Australia, for example, are either absent or poorly represented in the south-west (Bunn and Davies, 2000). The coastal draining rivers flowing east from the Eastern Highlands are also generally short, but with high-energy, steep headwater sections (Schofield *et al.*, 2000).

Tasmanian rivers are geologically recent and many with steep slopes and high discharges are actively eroding (White, 2000). Unlike the mainland there is also widespread evidence of the influence of recent (15 to 22 thousand years ago) glacial and peri-glacial activity on river form: wide valleys, moraines and deposits of loosely packed boulders (Jerie *et al.*, 2003).

1.2. The distinctive character of Australian streams

Many authors have noted the distinctive character of Australian river systems (Williams, 1980; Dunn, 2000; White, 2000; Brierley and Fryirs, 2002). Rainfall and run-off patterns that are consistent with the usual concept of the hydrologic cycle are found only in Tasmania and the coastal margins of the Australian continent (White, 2000). Most streams are intermittent or episodic. Very few are affected by snowmelt. For many large inland rivers, the lack of substantial tributary inflows and high rates of infiltration and evaporation reverse the usual

discharge –area relationships. Transmission losses on Cooper Creek, for example, can exceed 75% (Knighton and Nanson, 1994a) and few floods reach the terminal Lake Eyre (Kotwicki, 1986).

Extreme variability of runoff is characteristic of most inland rivers. Extensive flooding may follow long periods of little or no flow but such floods occur erratically (Kingsford, 2002). The variability of flow in Australian streams (measured as the median value of the coefficient of variation of annual flows) is more than double that of Europe, Asia and North America (McMahon, 1979). Cooper Creek and the Diamantina River are perhaps the most variable of all rivers globally (Puckridge *et al.*, 1998). A unique characteristic of this variability is the fact that it tends to increase with increasing size of the catchment (Lake, 1995).

Much of the Australian continent has very low relief, a result of its great age, relative tectonic stability and millions of years of accumulation of deposited material (Young, Schiller, Roberts *et al.*, 2001). Average slopes are lower than on any other continent (Vörösmarty *et al.*, 2000). Large floodplain rivers have such low gradients over much of their length that travel times for floodwaters are measured in weeks, or even months (e.g. Knighton and Nanson, 2001). Many geomorphic processes and the river forms they create are rarely seen elsewhere (White, 2000; Brierley and Fryirs, 2002). Only in the areas of higher relief do channel networks match the typical textbook descriptions of a connected, dendritic drainage network. There are surprisingly few truly alluvial self-adjusting rivers (Brierley and Fryirs, 2002) while anabranching rivers, globally uncommon (Knighton, 1998), occur in many parts of the continent (Knighton and Nanson, 1994b; Wende and Nanson, 1998; Tooth and Nanson, 1999; Young, Schiller, Harris *et al.*, 2001; Tooth and Nanson, 2004; Judd *et al.*, 2005). In the south-west the usual patterns of drainage network evolution are reversed. While the headwaters are usually the most actively eroding sections of a river system, in the south-west the lower sections of rivers on the geologically young coastal plains and Darling Range cut back into the remnants of ancient palaeochannels on the Yilgarn Plateau (Pen, 1999).

Australian river systems are distinguished biologically by the occurrence of several groups not found, or at least not common, outside of Australia (Williams, 1980). Almost all invertebrates (Lake, 1995) and most freshwater crayfish (Whiting *et al.*, 2000) are endemic. Numbers of fish species are relatively low, perhaps explained by the extreme variability in discharge (Lake, 1995) but especially in the drier regions there are high levels of endemism in this group as well (Unmack, 2001). Adaptive radiation has produced greater diversity among several groups (e.g. some families of stoneflies, mayflies, caddisflies and freshwater crayfish) than elsewhere (Lake, 1995; Boulton and Brock, 1999; Whiting *et al.*, 2000). Endemism and Gondwanic affinities are evident among riparian plants and aquatic macrophytes (Dunn, 2000) while the platypus, one of only three species of monotreme in the world, is found only in the river systems of eastern Australia (Grant, 1984).

1.3. The conservation imperative

Rivers have been the focus of life for the indigenous peoples of Australia for thousands of years. Archaeological evidence suggests human association with the Paroo River in north-western New South Wales and its lakes and floodplains extending at least 14,000 years (Robins, 1999). Rivers are central to the identity of many indigenous peoples (Langton, 2002; Behrendt and Thompson, 2003). They are a favourite place for camping and hunting, gathering and fishing (Behrendt and Thompson, 2003) and the focus of creation stories (O'Connor *et al.*, 1989; Hope, 2004). The paths that linked places of significance often followed stream courses and were kept open by periodic burning (O'Connor *et al.*, 1989). Indigenous people have extensively utilised the resources of the rivers (Hope, 2004). Remains of extensive networks of canals and complex systems of fish and eel traps have been found in rivers in many parts of the continent (O'Connor *et al.*, 1989; Bandler, 2002). Hope (2004) reports the finding of the remains of several hundred golden perch apparently caught in sophisticated gill nets in a single event. Barmuta (2003) also cites reports by Gilmore (1934) of the routine translocation and husbandry of fish.

The strong connection of indigenous people with place imposes obligations to respectfully manage those places (Roberts and Wallis, 2003). Cultural mechanisms have evolved to protect important resources. By placing restrictions on the use of special cultural places and sacred areas, such as the Rainbow Serpent sites in the deep pools in the river, indigenous groups created refuges for fish and other aquatic biota (Behrendt and Thompson, 2003). Traditional burning practices ensured abundant food and other resources but also promoted biological diversity and maintenance of rare and range restricted fauna (Yibarbuk *et al.*, 2001). Frequent and regular burning, however, changes the composition and structure of vegetation (Bond and Keeley, 2005) and alters patterns of runoff and sedimentation to rivers (Olsen and Skitmore, 1991). Debate over the role of the first humans in the extinction of the Australian megafauna is growing (Burney and Flannery, 2005; Miller *et al.*, 2005) with recognition of profound changes to terrestrial ecosystems following human arrival (Johnson, 2005). These changes must have affected catchment processes, but the evidence for direct impacts on rivers is limited (Pickup, 1986).

Overall, little is known of the impacts of indigenous peoples on aquatic systems in Australia, but it is usually accepted that they are far less profound than those that followed European colonization in 1788 (Barmuta, 2003). Certainly, the pace of change has accelerated dramatically over the last 200 years. Serious land and water degradation occurred within the first few decades of European settlement (Wasson *et al.*, 1998; Abernethy *et al.*, 2005). Starr (1999), for example, presents dramatic evidence of extensive gullying that occurred almost immediately after the introduction of grazing by hard-hoofed animals in the upper Murrumbidgee River catchment. Drainage lines that, at the time of European settlement were

swampy or chain-of-ponds systems, were rapidly incised and widened. Further erosion followed broad-scale clearing in the 20th Century.

Through the observations and trial and error of thousands of generations, indigenous peoples accumulated a detailed and localized ecological understanding (Horstman and Wightman, 2001; Rose, 2001). But with European colonization, indigenous knowledge and practices were marginalised (Goodall, 1999) and replaced by European-style land and water management practices characterized by “Mis-reading, inappropriate use and unrealistic expectations” (White, 2000 p.1). The serious consequences for Australian river systems are well illustrated in recent State of the Environment reports (State of Environment Advisory Council, 1996; Ball *et al.*, 2001). These reports describe the excessive water extraction, pollution, eutrophication and algal blooms, catchment modification, habitat destruction and flow regulation that have substantially altered and degraded habitat quality, especially in lowland rivers and agricultural catchments. They record serious declines in the range and abundance of many native aquatic species while exotic species such as carp, trout and Giant Sensitive Plant (*Mimosa pigra*) have flourished. Many native fish species are now threatened or endangered. Among the 200 or so freshwater species in Australia, 11 are listed as endangered and 10 as vulnerable. Among bird species, waterbirds show the most consistent declines (Olsen and Weston, 2004). In many of the major wetlands dependent on river flows, numbers of waterbirds have shrunk dramatically (Kingsford and Thomas, 1995, 2004; Olsen and Weston, 2004). For many other groups including platypus and crayfish, numbers appear to be declining but there is little reliable information on their distribution and abundance (Ball *et al.*, 2001).

River degradation is extensive and widespread. In a national survey of river health using the AusRivAS bioassessment method (Coysh *et al.*, 2000), 40% of sites were found to be impaired to some degree and 9% severely or extremely degraded (Ball *et al.*, 2001). There may be significant lag effects however, so the full influence of catchment activities on river systems may not be evident for many decades (Frissell and Bayles, 1996; Harding *et al.*, 1998; Boulton and Brock, 1999). Dryland salinity, for example, a result of the replacement of the deep-rooted native vegetation with shallow-rooted pastures and crops, is expected to be the major contributor to increased salinisation of rivers over the next 100 years (Ball *et al.*, 2001). Not surprisingly, the worst impacts occur within the intensive landuse zone, especially in the southeast and southwest of the continent. The Assessment of River Condition (Norris *et al.*, 2001) found that over 85% of the 210,000km of river length assessed within this zone, occupying about 40% of the landmass, was significantly modified from the original condition by the aggregate effects of resource use; some 19% of this was classed as substantially modified. Large rivers in particular, suffer the cumulative effects of human activities upstream with only 2% of large rivers (catchment area > 5000km²) unaffected by some form of human disturbance post-European settlement (Stein *et al.*, 2002).

The threats to river systems are ongoing and in many cases expanding. Water use has dramatically increased over the last two decades putting many river systems under severe stress (Australian State of the Environment Committee, 2001; National Land and Water Resources Audit, 2001b). In the early 1990's, the major wetland system on the lower Murrumbidgee River, the Lowbidgee floodplain, was a candidate for listing as a wetland of international importance under the Ramsar Convention but by the end of the decade it was in serious decline due to substantially increased water resource development upstream (Kingsford and Thomas, 2004). More than a quarter of surface water management areas are close to, or have exceeded, sustainable extraction limits (National Land and Water Resources Audit, 2001b). Groundwater usage is also increasing, particularly where more restrictive controls on surface water use are imposed (Ball *et al.*, 2001) with potentially serious impacts on many river systems dependent on groundwater inputs to sustain base flows (Nevill and Phillips, 2004). Pressure is now growing to develop the relatively unimpacted tropical rivers in the north (Arthington *et al.*, 2004; Kingsford *et al.*, 2005).

1.4. Conservation initiatives

Australia's globally significant river ecosystems are seriously threatened and urgently in need of conservation. Their health is cause for concern and anger among indigenous people, due they believe, to the lack of a holistic and respectful approach to the land and its resources (Forward NRM and Arrilla – Aboriginal Training & Development, 2003). Indigenous groups now seek a more substantive involvement in planning and management and recognition of their water rights (Morgan *et al.*, 2004). For several decades, probably longer, concerned scientists and community groups have been calling for conservation action (e.g. Williams, 1967; Lake, 1980; Macmillan, 1984; Barmuta *et al.*, 1992; Dunn, 2000; Schofield *et al.*, 2000; Yencken and Wilkinson, 2001; Georges and Cottingham, 2002; Barmuta, 2003; Wentworth Group of Concerned Scientists, 2003; Arthington *et al.*, 2004; Nevill and Phillips, 2004). Yet despite mounting evidence of the pervasive effects of human activities, river conservation still lags well behind that for terrestrial and more recently, marine environments (Schofield *et al.*, 2000; Cullen, 2003; Nevill and Phillips, 2004; Kingsford *et al.*, 2005).

Responsibility for land and water management rests primarily with the States and Territories. Victoria leads the way in river protection programs. Assessment of the ecological value of Victorian streams during the 1980's (Macmillan, 1984; Macmillan *et al.*, 1987; Kunert and Macmillan, 1988; Land Conservation Council, 1989) led to the designation of 15 Representative Rivers (Land Conservation Council, 1991) and legislative protection for 18 Heritage River reaches and 26 small "Essentially Natural Catchments" (Bennett and Doolan, 2004). New initiatives promise conservation measures for river systems in other states. In September, 2005 the Queensland parliament passed legislation to protect "wild rivers". Eighteen relatively

undisturbed rivers were nominated for legislative protection (Cordner, 2004; Amprimo, 2005). Tasmania is developing a conservation plan for freshwater systems based on the principles of comprehensiveness, adequacy and representativeness (Long, 2005). Legislation or policies applicable to the protection of freshwater ecosystems already exist in all states (Nevill, 2004). New South Wales, for example, has fisheries legislation providing for the establishment of aquatic reserves (Hankinson and Blanch, 2002; Hankinson and Blanch, 2003) and the National Parks and Wildlife Act has provision for the protection of wild and scenic rivers (Conlon, 2004). Nevill (2004), however, argues that while there are adequate policy and legislative settings in place there has been little commitment to strategic implementation. For example, management plans for Victoria's Heritage Rivers are still incomplete, more than a decade after their declaration (Nevill and Phillips, 2004). In New South Wales no aquatic reserves have been declared in freshwater systems (Hankinson and Blanch, 2003) and no Wild and Scenic rivers were listed in the 18 years since the provisions were enacted, although six were recently proposed (Conlon, 2004). Less than 2% of the area of floodplain wetlands is protected in the State's reserve system (Kingsford *et al.*, 2004). NSW has no state-wide program for assessing riverine conservation value and applies no consistent assessment method creating "a patchwork of data and evaluations, sometimes in apparent contradiction of one another" (Chessman, 2002 p.2).

Reserves, areas set aside to protect their natural values, are the foundation of a conservation strategy (Margules and Pressey, 2000; Meir *et al.*, 2004). Australia was one of the early leaders in the development of a reserve system. Royal National Park near Sydney was declared in 1879 not long after declaration of the world's first national park, Yellowstone National Park in the United States (NSW National Parks and Wildlife Service, 2004). By 2002, reserves or protected areas occupied an area of 775,000 km² or about 10% of the continent (Environment Australia, 2003). Many reserves however, were chosen because of their scenic or recreation attributes or low value for commercial uses (Pressey, 1994; Nix, 1997). Rarely was conservation of rivers stated as a reason for reservation (Lake, 1980; Nevill and Phillips, 2004).

There has been no systematic assessment of the adequacy of the existing protective measures for riverine ecosystems (Nevill, 2004). Clearly, many reserves provide inadequate protection for rivers and their dependent ecosystems because reserve managers have no control on water use or catchment activities outside the reserve (Kingsford and Thomas, 1995; Jerie *et al.*, 2001; Cullen, 2003). Few reserves encompass an entire river basin. Even within protected areas, the detrimental effects of activities such as the discharge of sewage effluent, stocking of exotic fish for recreation and water use for resorts and snow making, are not seriously recognised (Cullen, 2003). In fact, nearly half of all streams within existing nature conservation reserves have been disturbed to some extent by current or past activities within the reserve or upstream (Stein *et al.*, 2001).

It is likely that many of Australia's riverine ecosystems and their biota are poorly represented in existing protected areas. The reserve network favours steep and rugged terrain or remote or less productive areas and does not necessarily protect the most threatened systems (Pressey *et al.*, 2000). Thus, Tait *et al.* (2003) suggested that the existing reserve network was biased towards higher gradient streams in upper catchment areas and protected very few lower gradient mid-catchment and lowland floodplain systems. They also found the reserve network poorly represents Unmack's freshwater fish bioregions (Unmack, 2001). For about half of the 31 bioregions less than 5% of the stream length fell within a reserve (Tait *et al.*, 2003).

The National Reserve System (NRS) program was established in 1992 out of concerns (e.g. Specht, 1975) about the extent to which the State and Territory systems of protected areas were representative of the terrestrial ecosystems of Australia (Nevill and Phillips, 2004). The program is committed to the establishment of comprehensive, adequate and representative systems of reserves (Commonwealth of Australia, 1999). It operates together with the Regional Forest Agreement (RFA) process for the conservation of the forest estate (JANIS (Joint ANZECC / MCFFA National Forest Policy Statement Implementation Sub-committee), 1997) and the National Representative System of Marine Protected Areas (NRSMPA) (<http://www.deh.gov.au/coasts/mpa/nrsmpa>) (CSIRO Division of Fisheries and CSIRO Division of Oceanography, 1996).

The NRS program has neglected freshwater conservation values. Areas for inclusion are selected using the national and regional planning framework provided by the Interim Biogeographic Regionalisation for Australia (IBRA) (Thackway and Cresswell, 1995; Environment Australia, 2000) but this landscape regionalisation may not adequately reflect the distribution patterns of freshwater biodiversity (Wells and Newall, 1997; Marchant *et al.*, 1999; Turak *et al.*, 1999). In particular, it does not recognise the catchment-based constraints on obligate freshwater taxa (Tait *et al.*, 2003). Thus, Nevill (2004; Nevill and Phillips, 2004) and others (e.g. Dunn, 2000; Tait *et al.*, 2003) argue for development of a classification tailored to the needs of freshwater systems to support their inclusion within a comprehensive, adequate and representative network of protected areas.

Other programs were developed to protect rivers with special values outside of protected areas. For example, lists of wild rivers (rivers largely unaltered by European settlement) (<http://www.heritage.gov.au/anlr/code/arc-maps.html>) (Stein *et al.*, 2001) and accompanying conservation management guidelines (Environment Australia, 1998) were produced in collaboration with State and Territory agencies. Unfortunately, national programs encouraging adoption of the management guidelines never eventuated.

Australia endorsed the Ramsar convention, an international convention that requires member countries to nominate and manage wisely, wetlands of international importance (Ramsar Convention Secretariat, 2003). The convention defines wetlands broadly to include rivers and

streams, but in Australia it has only been applied to lentic (standing water) systems, consistent with the Australian usage of the term (Nevill and Phillips, 2004). Because of uncontrolled water resource developments upstream, Ramsar listings have often been similarly ineffective at protecting wetlands (Kingsford and Thomas, 1995; Cullen, 2003; Fisher, 2004; Fazey *et al.*, 2006). Furthermore, wetlands that satisfy the criteria for a “wetland of international importance” may not necessarily hold the highest conservation value because the Ramsar criteria do not require any systematic comparison of values among all wetlands (Kingsford *et al.*, 2005).

1.5. Developing a conservation plan for Australian rivers and streams

Many other conservation initiatives across all levels of government are also helping to conserve Australian river ecosystems, but there is no overarching plan to co-ordinate these programs or ensure they are targeted towards the most urgent needs. Thus, even where effectively implemented, these initiatives may not be maximizing the conservation benefits that flow from the use of scarce conservation resources.

A conservation plan would enable a range of conservation measures (reservation, land-use planning and management of threats) to be strategically employed and integrated within the landscape (Dunn, 2000; Yencken and Wilkinson, 2001; Kingsford *et al.*, 2005). It would include effective strategies for the management of the cumulative effects of incremental development (Nevill, 2003) and set out systematically identified priorities for conservation action, focused on ecosystem functions and processes and the habitat they provide (Kingsford *et al.*, 2005). Comprehensive and integrated planning, however, requires a national or continental perspective.

1.5.1. A continental perspective

Nationally coordinated conservation plans conserve biodiversity more efficiently. They require less area (Pressey and Nicholls, 1989a; Erasmus *et al.*, 1999) and associated costs are lower (Strange *et al.*, 2006) than equivalent, independently derived, regional strategies. A continental framework for conservation planning ensures that conservation actions are coordinated across jurisdictional boundaries and thus the benefits accrue more widely (Corkum, 1999; Jensen, Christensen *et al.*, 2001). Parochial decision-making can undermine environmentally sustainable management, especially for cross-border rivers (Kingsford *et al.*, 1998; Australian State of the Environment Committee, 2001; Kingsford *et al.*, 2005). A national framework makes it harder for State or Territory interests to ignore broader conservation objectives and guides the priority setting of national funding programs.

The relative conservation value of any river (e.g. locally, regionally or nationally important) is only apparent from a continental assessment. A continental assessment provides unambiguous evidence of the cumulative effects of impacts (Boulton and Brock, 1999; Ormerod, 1999; Frissell *et al.*, 2001) and hence the need for conservation actions, often underestimated by local or regional studies (Hughes *et al.*, 2000; Australian State of the Environment Committee, 2001). It sets the context for more detailed and specifically targeted assessment and planning (Veitch and Walker, 2001).

Consistent information on biodiversity is necessary to compare the conservation value of potential areas in order to identify priorities for conservation action (Margules and Pressey, 2000). A continental planning framework avoids the artificial constraints imposed by administrative boundaries. For meaningful interpretation or modelling of distribution patterns the assessment area should cover the geographic and environmental range of most of the species of interest (Austin and Margules, 1986; Nix, 1986; Hunter and Hutchinson, 1994; Bourgeron *et al.*, 2001c; Pausas and Austin, 2001; Gillison and Liswanti, 2004). At geopolitical scales smaller than continental, the range of many taxa are likely to be truncated. Cross jurisdictional information on the range of spatial variation within and among river types is also essential to inform effective river classification (Kingsford *et al.*, 2005)

1.5.2. Systematic conservation planning

Systematic conservation planning provides a structured framework to locate priority areas for conservation action (Box 1.1). Essential preparatory tasks include consideration of the opportunities and constraints for implementation (policy, legal, institutional and socio-economic) and establishing the involvement of key stakeholders (Cowling and Pressey, 2003). The approach has been widely applied in terrestrial and to a lesser extent, marine environments although only recently introduced to the conservation of riverine ecosystems (Roux *et al.*, 2002; Smith *et al.*, 2002; Weitzell *et al.*, 2003; Pressey, Albernaz *et al.*, 2004; Sowa *et al.*, 2004; Linke *et al.*, 2005; Long, 2005).

Conservation planners usually have a vision of desired conservation outcomes that is specified in relatively general terms for ease of communication (e.g. Pressey *et al.*, 2003). Systematic conservation planning requires this vision be translated into specific goals, specifying the features that are the focus of the conservation plan, and measurable targets. Conservation of biodiversity is the widely adopted objective of many international conventions, national governments, state agencies and non-government organizations (Redford *et al.*, 2003). Yet, data are never available to fully describe and map the multiple levels of biological organization (genes, taxa, communities, ecosystems etc.) which makes up “biodiversity” so the conservation goals focus on surrogates (Margules *et al.*, 2002). Biodiversity surrogates may include species,

Box 1.1. Stages in systematic conservation planning (after Margules and Pressey, 2000).

- 1. Compile data on the biodiversity of the planning region and attribute to planning units**
- 2. Identify conservation goals and specify quantitative, operational targets**
- 3. Review existing conservation areas**
- 4. Select additional conservation areas**
- 5. Implement conservation actions**
- 6. Maintain the required values of conservation areas**

either all known species and/or with priority given to species that are endemic, rare or of special concern, indicator species (Simberloff, 1998; Poiani *et al.*, 2001; Kerley *et al.*, 2003), communities, ecological and/or evolutionary processes or ecosystems (Groves *et al.*, 2000; Redford *et al.*, 2003).

Specific, preferably quantitative, targets make conservation goals operational (e.g. Pressey *et al.*, 2003). Targets, a defining characteristic of systematic conservation planning, provide accountability and defensibility for conservation decisions and allow the contributions of existing protection mechanisms to be reviewed (Margules and Pressey, 2000). Assessment of the extent to which conservation targets are met by existing conservation reserves (e.g. Oldfield *et al.*, 2004; Rodrigues *et al.*, 2004; Sowa *et al.*, 2004) is an important step as it sets the context for the selection of additional areas to be added to the reserve system (Margules and Pressey, 2000).

Reserve selection algorithms identify a minimum set of areas or “planning units” that collectively satisfy conservation targets (Andelman and Willig, 2002; Pressey, 2002; Onal, 2004). They provide flexibility by facilitating consideration of alternative reservation configurations and trade-offs with competing land-uses (Margules and Pressey, 2000; Margules *et al.*, 2002). The algorithms operate on a data matrix of potential areas by biodiversity features, expressed either as a presence only or some measure of abundance and employ a measure of complementarity to select areas that contribute most unrepresented biodiversity features to an existing set of areas (Margules *et al.*, 2002). They have been adapted to minimise opportunity costs (e.g. foregone timber production) (Faith, Margules and Walker, 2001), account for spatial constraints (e.g. areas already committed or that must be excluded, reserve design considerations) (Lewis *et al.*, 1991) and integrate the knowledge of local and technical experts to overcome the limitations of available spatial data (Balram *et al.*, 2004).

The implementation phase of systematic conservation planning is more problematic (Margules and Pressey, 2000; Meir *et al.*, 2004; Pressey, Watts *et al.*, 2004). Where the available resources require staged implementation, a strategy for scheduling action must be devised to minimize the extent to which conservation targets are compromised before the plan is fully implemented (Margules and Pressey, 2000). Pressey and Taffs (2001), for example, argue that conservation benefits are maximized by prioritizing action according to the criteria of vulnerability, that is, the likelihood or imminence of loss of the conservation values of an area and irreplaceability, being the likelihood that an area will be needed to contribute to the nominated set of conservation targets.

Systematic conservation planning procedures provide the basis for explicit, effective and accountable decisions about the allocation of limited conservation resources (Pressey and Cowling, 2001). They consider the contribution to the conservation targets of all areas consistently and systematically. The systematic planning framework focuses on the dual objectives of representativeness, the need to represent the full range of biodiversity and persistence, the long-term retention of biodiversity (Margules and Pressey, 2000). Other conservation planning approaches work with different perceptions of conservation value (Whittaker *et al.*, 2005). Conservation International's Global Hotspots Program (Myers *et al.*, 2000) places highest priority on the centres of endemism under greatest threat, thereby emphasising the conservation of range-restricted species (Whittaker *et al.*, 2005). Criteria such as naturalness, importance for certain life history stages, evolutionary significance, diversity and rarity are also widely adopted to identify conservation importance (Boon *et al.*, 1998; Bennett *et al.*, 2002; Dunn, 2003). While there is ongoing debate about the merits of these different approaches (e.g. Pressey and Nicholls, 1989b; Araújo and Williams, 2001; Myers, 2003; Dimitrakopoulos *et al.*, 2004; Kati *et al.*, 2004) there is little doubt that the defining characteristics of systematic conservation planning (i.e. transparency, repeatability, based on explicit goals and consistent and systematic application) are generally applicable. Systematic conservation planning methods may also strategically guide the placement of conservation measures outside of reserves (Pressey and Logan, 1997) including the implementation of economic instruments (e.g. incentives, levies and biodiversity credits) for biodiversity conservation (Faith *et al.*, 2003).

1.5.3. An ecosystems approach

Conservation practice has shifted from a focus on individual iconic species to one that emphasizes conservation of functioning ecosystems including the biota and the dynamic, multi-scale ecological patterns and processes of the natural systems that sustain them (Noss, 1996; Simberloff, 1998; Poiani *et al.*, 2000; Rowe, 2001; Kingsford *et al.*, 2005). An ecosystems

approach to river conservation recognises that rivers are much more than the linear channel represented by the blue lines on a map. Surface waters, sub-surface waters, riparian / floodplain systems and associated processes are all integral components of riverine landscapes or “riverscapes” (Ward, 1998). Upland streams are tightly connected with their catchment, their character and function strongly influenced by catchment conditions (Young, Schiller, Harris *et al.*, 2001) while floodplains and the swamps, channels, lakes, billabongs, wetlands and waterholes they contain, are the dominant feature of large lowland rivers (Kingsford, 2002). Energy, materials and biota are transferred via linkages operating in multiple dimensions: longitudinal (up/downstream), lateral (channel/ margins) and vertical (surficial/groundwater) (Ward, 1998; Piégay and Schumm, 2003; Baxter *et al.*, 2005). These linkages sustain ecological processes in river corridors (Ward *et al.*, 2001) and create a dynamic, complex mosaic of smaller scale aquatic ecosystems (Amoros and Bornette, 2002) arrayed hierarchically (Frissell *et al.*, 1986). Biological diversity results from interactions between processes operating at various spatial and temporal scales (Lammert and Allan, 1999; Amoros and Bornette, 2002; Lamouroux *et al.*, 2004). Variation in the nature and intensity of hydrological connectivity, for example, is an important driver of the heterogeneity of floodplain ecosystems (Kingsford *et al.*, 1998; Sheldon *et al.*, 2002; Arthington *et al.*, 2005; Capon, 2005).

1.6. A continental landscape framework to support systematic conservation planning

The task of preparing a continent-wide conservation plan with systematically and consistently identified priorities for conservation action, is critically dependent on the availability of a suitable spatial framework and a classification of rivers (Kingsford *et al.*, 2005). A spatial framework delineates the spatial units to be conserved and defines the geographic limits of the planning domain. Classification, a process for distinguishing different types of rivers, is important for a range of conservation planning and assessment tasks (Boon, 1992; Dunn, 2000; O’Keefe and Uys, 2000) and in particular, is essential for evaluating the criteria of representativeness (Austin and Margules, 1986). The spatial framework and classification are fundamental components of a continental landscape framework to support systematic conservation planning yet there is neither a national classification of river types in Australia (Schofield *et al.*, 2000) nor a suitable spatial framework.

1.6.1. Research aims

This research aims to address these gaps to support a systematic and co-ordinated national approach to conservation planning for Australian rivers and streams. Specifically it seeks to:

- develop a continent-wide spatial framework applicable to the distinctive nature of Australian river systems,
- develop and evaluate a landscape classification for Australian rivers, and
- demonstrate the utility of the landscape framework for conservation planning

1.6.2. Scope

“Rivers” is used here in the broader sense of river ecosystems or riverscapes as described in Section 1.5.3. The landscape framework will embrace large rivers and small streams (or “creeks” in Australian terminology) whether permanent, ephemeral, intermittent or seasonal. Following Dunn (2000), it also includes ecosystems dependent on flows from the river including complex and effluent channels, terminal and floodplain wetlands and riparian communities. Estuaries, while influenced by river processes, are excluded as they would also require analysis of tide and wave processes (Heap *et al.*, 2001), a task beyond the scope of this project.

1.6.3. Thesis structure

The parts of this thesis are organised around its three principle aims. It begins in Part 1 with the development of a continent-wide spatial framework. Chapter 2 reviews the conservation literature to build understanding of the desirable characteristics of spatial frameworks and their function in the systematic conservation planning process. It then develops the theoretical structure for a spatial framework to serve continental-scale conservation planning. Chapter 3 compares existing spatial frameworks with Australia-wide coverage to this theoretical structure. Finding the existing spatial frameworks to be inadequate this chapter next reviews the methods that might produce a new framework, raising issues about their suitability for continent-wide application and ability to represent the distinctive drainage structures found in Australia. Accordingly, Chapter 4 sets out a new method and describes its application to derive an improved spatial framework for Australian rivers. This chapter necessarily includes substantial technical content. Chapter 5 evaluates the Pfafstetter system, an efficient system for delineating and coding hydrological units that has been adopted internationally.

Part 2 considers the task of classification beginning in Chapter 6 with a review of established classifications and the approaches used to derive them. This review concludes that the landscape classification approach exemplified by the environmental domain method is an option for continental scale river classification for conservation planning applications. Chapter 7 examines this conclusion. It details the development and classification of a database describing the landscape characteristics of the individual units of the spatial framework. Different ways of structuring the classification are tested and evaluated with a range of ecological site data in Chapter 8.

Part 3 describes the application of the landscape framework components, developed in parts one and two, to conservation planning. Chapter 9 demonstrates their use with a review of the existing protected area system, an essential stage in systematic conservation planning (Box 1.1). The thesis concludes (Chapter 10) with discussion of the outcomes of this research and its implications for conservation planning and assessment. It explores how this research might address the conservation planning needs of Australian rivers and streams and then examines its contribution to conservation planning activities elsewhere.

Chapter 2. Defining a spatial framework

Chapter 1 identifies the spatial framework as a key element underpinning a systematic, national approach to the conservation of rivers and streams. This chapter explores the role of the spatial framework and the types of spatial units employed with examples from the conservation literature (Section 2.1). These include both the terrestrial and aquatic (marine and freshwater) realms. Not surprisingly, it finds catchment-based units to be the most widely adopted spatial units for river conservation planning. The selection of a spatial framework is a key decision in the planning process and has implications for both conservation assessment and implementation. Issues associated with the selection of an effective spatial framework are discussed in Section 2.2. It is shown how the choice of spatial scale and recognition of the multiple dimensions of connectivity may influence conservation outcomes. The need for a multi-scale spatial framework becomes clear. In fact, riverine systems comprise a natural nested hierarchy of spatial units. Each of these hierarchical levels is evaluated in Section 2.3 in order to develop the theoretical structure for a hierarchical, nested spatial framework for continental scale conservation planning and assessment.

2.1. Spatial frameworks and conservation planning

Defining a suitable geographic framework is one of the first steps in a conservation planning exercise (Sowa *et al.*, 2004). It identifies the boundaries of the assessment area and delineates the geographic units into which it will be sub-divided for evaluation and prioritization for conservation action.

A wide variety of geographic units have been employed for conservation planning and assessment, sub-dividing assessment areas that range in extent from small sub-catchments (e.g. Scientific Panel for Lower North Coast River Management Committee, 1999) to large continents (Master *et al.*, 1998; Abell *et al.*, 2000). The size and/or configuration of selection units are usually different to that of the target features (Pressey and Logan, 1998). Grid cells or other regular shapes are easily delineated and have been a frequent choice (Kirkpatrick, 1983; Margules and Stein, 1989; Belbin, 1993a; Nicholls and Margules, 1993; Kirkpatrick and Brown, 1994; Sarakinos *et al.*, 2001; Cowling *et al.*, 2003; Lombard *et al.*, 2003; Dimitrakopoulos *et al.*, 2004). Grid cells were used for one of the earliest applications of a reserve selection algorithm (Pressey, 2002). In this case, Kirkpatrick (1983) used the 1km² grid cells delineated on the 1:100,000 scale topographic mapsheets as selection units to identify areas that should be added to the reserve network to protect rare Tasmanian plants. Where cadastral boundaries are mapped, tenure parcels may also be a suitable planning unit (McKenzie *et al.*, 1989; Pressey and Nicholls, 1989a).

In fragmented landscapes the choice of selection units is often dictated by landuse. Margules and Nicholls (1987) for example, used remnant patches of mallee vegetation on the Eyre Peninsula in South Australia to identify the set of patches that would have an acceptable likelihood of representing each plant community. Conservation assessments for habitats that occur in naturally discrete patches such as wetlands also employ selection units defined by the occurrences of the habitats concerned (Pressey and Logan, 1998). Beck and Odaya (2001) found that the bays and estuaries of the nearshore waters delineated natural units that were then used to prioritize sites for conservation in the Gulf of Mexico ecoregion. Similarly, to identify the minimum set of wetlands needed to represent all wetland plant species on the Macleay Valley floodplain, Margules *et al.* (1988) used wetland boundaries to define the planning units.

Planning units may be a combination of both regular and irregular units. The Great Barrier Reef Marine Park Representative Area Program treated individual reefs as separate units and divided non-reef areas into 10km² or 30km² hexagons, delineating over 16,000 planning units for the 343,500km² of the park (Day *et al.*, 2002). The size of the hexagon unit depended on the level of information available; smaller units were used in areas where more information was available. The planning units provided the spatial framework for the application of a reserve selection algorithm to identify areas for re-zoning as “no-take” zones to protect representative examples of the habitats and communities of the park (Day *et al.*, 2003).

One of the most comprehensive conservation planning programs to date is that undertaken for the Cape Floristic region at the south-western tip of Africa, a globally recognized region of high endemism and diversity (Cowling and Pressey, 2003). The plan sets targets for protection for five groups of features: broad habitat types; plant species in the family Proteaceae; reptiles, amphibians, freshwater fish, large and medium-sized mammals and spatial surrogates for ecological and evolutionary processes (Pressey *et al.*, 2003). The areas that would be important to achieving these targets were selected from a set of planning units that included the configuration of specified process components (edaphic interfaces, riverine corridors, sand movement corridors, and upland–lowland interfaces), sixteenth degree grid squares (about 40km²), statutory reserves and non-statutory reserve units (Cowling *et al.*, 2003). Non-statutory reserves, being of less certain protection status, were sub-divided into sections corresponding with the sixteenth-degree squares so that those parts that contributed most to achieving conservation targets could be selected. The planning domain for this study was centred on the Cape Floristic Region, extended by 60km beyond the original region boundaries to allow the conservation plan to link with adjacent regions (Cowling *et al.*, 2003). However, neither the basis for the 60km overlap nor the mechanisms for facilitating these important linkages were discussed.

Pragmatic decisions often dictate the choice of spatial units. The Resource Mapping Units (RMUs) chosen as planning units in the BioRap project (Biodiversity Rapid Appraisal Pilot

Project) for Papua New Guinea were already widely used and understood by Papua New Guinea government agencies (Nix *et al.*, 2000). RMUs are land areas with relatively homogenous topography, climate and lithology (Bellamy, 1986) for which a wide range of relevant data had been compiled. Over 4000 RMUs varying in size from less than 1 km² to 8,500 km² were used as planning units to systematically identify priority areas that would represent the biodiversity of Papua New Guinea. In this case, the planning domain reflected administrative borders - the project brief and the RMUs extended only to the Papua New Guinea administrated part of the island of New Guinea.

Systematic conservation planning approaches have only recently been applied to river ecosystems. Catchment based units are widely adopted. An assessment of 4706 catchment units was the basis for the recently developed list of New Zealand's Waters of National Importance (WONI), nationally important aquatic systems selected to represent the full range of New Zealand's indigenous riverine biodiversity (Chadderton *et al.*, 2004). The catchment units represented whole-river catchments or major tributaries, nested within larger (>1,000 km²) catchments. In the United States, critical areas for conserving populations of all freshwater fish and mussel species at risk were identified from a catchment based analysis (Master *et al.*, 1998). Abell *et al.* (2000) defined catchment-based ecoregional units for their assessment of freshwater conservation status for North America. Seventy-six ecoregions were delineated by aggregating catchments according to patterns of distribution of obligate freshwater taxa (fish, crayfish and mussels). Ecoregion units were prioritized for conservation action according to ratings for biological distinctiveness, based largely on species richness and endemism and conservation status, founded on indicators of degradation and alteration to natural river processes.

Catchment units may be further sub-divided according to topography and the topology of the drainage network. Phillips *et al.* (2002), for example, tested a new method of determining conservation value developed by the Queensland Environmental Protection Authority (Phillips *et al.*, 2001; Bennett *et al.*, 2002) for streams in the Burnett River basin in Queensland. They defined selection units, termed "conservation mapping units", by progressive sub-division of the basin into tributary catchments and mainstem sub-catchments. The level of sub-division adopted was chosen to maximize the coverage of the data to be used for evaluation within catchment units. A fourth level sub-division produced 567 conservation mapping units that ranged in size from 1km² to 600km² with an average around 60km². The length of stream included in these sub-catchment units also varied greatly, between 1 and 66 kilometres (average 11km).

The planning units employed need not be limited to a single spatial scale. Sowa *et al.* (2004), for example, undertook an assessment at the regional, catchment, and valley segment scales to identify gaps in the current conservation management efforts for aquatic biodiversity in Missouri. Similarly, the West Gippsland Catchment Management Authority adopted a hierarchical spatial framework to assist in the task of setting priorities for river management

activities (West Gippsland Catchment Management Authority, 2004). Environmental, social and economic asset and threat data were collected, valued and validated at both sub-catchment and river reach scales to determine highest value areas across the West Gippsland region of Victoria. A number of other factors in addition to topography, including land use, river form and process, local knowledge and internal and external operational and reporting requirements influenced the placement of spatial unit boundaries. Thus, for example, a major in-stream structure defines the boundary between upper and lower reaches in a number of sub-catchments.

SERCON (System for Evaluating Rivers for Conservation) (Boon *et al.*, 1998; Boon *et al.*, 2002) is perhaps the most established of the systems for assessing the conservation value of rivers. Developed in the United Kingdom, it also uses river reaches as assessment units.

Conservation values, specified in terms of conservation criteria including naturalness, representativeness, species richness, and rarity, are evaluated for discrete lengths of river, normally between 10 and 30 km known as Evaluated Corridor Sections (ECSs). Natural physical features such as channel gradient, catchment geology, lakes, and major tributary junctions are used to divide rivers. The resulting ECSs are considered to represent ecologically relevant units for evaluation (Boon *et al.*, 1997 cited in Boon *et al.*, 2002).

The Conservation of Freshwater Ecosystem Values (CFEV) project is a comprehensive program that aims to identify and prioritize for protection, areas of Tasmania with high conservation value for freshwater dependent ecosystems (Long, 2005). A high resolution (1:25,000 scale) stream network (Corbett *et al.*, 2005), sub-divided into over 300,000 individual stream segments, provided the basic spatial units for the riverine component of the CFEV database. A combination of empirical, modelled and other derived data were classified and attributed to stream segments to characterize key physical (geomorphology, network position and hydrological regime), biological (fish, macroinvertebrate, riparian vegetation and macrophyte assemblages) and anthropogenic disturbance components of the riverine ecosystem, to facilitate the application of representativeness, distinctiveness and naturalness assessment criteria (Davies, 2005). These results, along with identification of other special conservation values (e.g. centres of endemism, important breeding sites) (Dunn and Heffer, 2005) enabled the ranking and grouping of the conservation value of river segments (Knight and Brown, 2005).

2.2. Choosing a spatial framework

Pragmatic considerations dictate the choice of spatial framework. Often, it will be limited by whatever data are at hand, because conservation needs are so urgent. There is little discussion of the merits of alternative spatial frameworks or the consequences of the choices made. Yet such decisions as defining the boundaries of the assessment area or the choice of planning units have important implications for conservation outcomes. A number of issues influence both the

process of assessment and the implementation of results. Foremost amongst these is that of scale.

2.2.1. Scale

Scale influences the modelling and analysis of hydrological, geomorphological and ecological patterns (Gallant, 1997). A change in scale changes the types of phenomena and processes that can be observed (Cousins, 1993) and the detection of ecological patterns (Wiens, 1989; Levin, 1992; Ward *et al.*, 2001). Stine and Hunsaker (2001) define two primary scaling factors; extent and grain size. The spatial extent or planning domain refers to the size of the area mapped or analysed and defines the context for the comparison of potential areas. Grain size is the resolution of any given landscape feature, as it is perceived through the source of data used (e.g. minimum mapping unit, pixel size).

Conservation value is often evaluated by application of thresholds and decision rules to specified criteria (Usher, 1986; Boon *et al.*, 1998; Dunn, 2000; Dunn, 2003). Commonly included criteria include naturalness, representativeness, rarity, uniqueness, richness and diversity. However, these criteria are sensitive to both the spatial extent and the grain size at which they are evaluated. Rarity, for example, is a measure of the relative occurrence of features. A rare feature is one that occurs at low density, or occupies a small geographic range (Whittaker *et al.*, 2005). Rarity is only defined with reference to a particular context. Thus, a stream type may be rare within a single catchment but common at a regional scale. Conversely, a stream type common in one catchment may be found nowhere else.

Spatial extent and resolution can dramatically shape perceptions of species richness patterns (see review in Rahbek, 2005). Stoms (1994) for example, found clear indications of scale effects in predicted patterns of gamma or landscape richness. He concluded that smaller spatial units (less than 100km²) sampled within-habitat (alpha) diversity rather than gamma richness and produced greater uncertainty. On the other hand, very large spatial units (greater than 1,000km²) that sampled most of the available habitats, smoothed out species richness differences between areas to the extent that they would not be useful to identify conservation priorities.

Grid cells or other regular sized planning units make the application of conservation criteria simpler. Unlike units based on natural boundaries such as catchments, they require no standardization to remove the confounding effects of variable sampling area (Whittaker *et al.*, 2005). However, where assessment units are delineated to represent relatively homogeneous sections of a river system, these effects on conservation evaluation may not be significant. Thus, for example, while the length of 120 ECS's used to compute SERCON scores varied greatly, ECS length was only weakly correlated with scores for rarity and species richness (Young Associates, 1999).

Recent studies suggest the scale of analysis also significantly affects the application of priority setting algorithms. They show that the spatial extent of the planning domain can have a very large impact on the minimum set of areas selected to achieve conservation targets (Erasmus *et al.*, 1999; Diamond *et al.*, 2005). Erasmus *et al.* (1999), for example, found very little overlap in the areas selected as important for conservation of mammal species across mainland sub-Saharan Africa when analysed at different geopolitical scales; local (within province) and regional (across provinces). Different planning units alter the patterns of occurrence of biodiversity features and hence the systematic selection of conservation priority areas (Warman *et al.*, 2004). However, it is unclear to what extent priority areas selected with smaller planning units are nested within those located using coarser scale units within the same geographical area. Comparing the results from priority analyses performed with the same data set compiled at 1, 2, 4 and 8 degree grid cell scales, the minimum sets required to represent each of the sub-Saharan Africa mammal species identified at finer scales were largely captured by the minimum sets selected at coarser scales (Larsen and Rahbek, 2003). In contrast, Warman *et al.* (2004) found little similarity between the minimum sets obtained with different planning units (hexagons of three different sizes and fine scale vegetation mapping units). While Rouget (2003) found considerable overlap in the conservation plans produced using broad and fine scale planning units, irreplaceability values, which measure the relative importance of the contribution of a planning unit to the representation targets, were much higher at the fine scale. Further research is required to enable useful generalizations on the effects of the size of the planning unit on the selection of priority areas.

The scale of analysis also has implications for the design and management of reserves (Margules and Pressey, 2000). The planning domain must be large enough to ensure protected areas can be selected and managed to minimize the negative effects of boundary conditions (Stanley and Boulton, 2000) and to understand the scale-dependent effects of human activities on aquatic ecosystem integrity (Roth, Allan *et al.*, 1996; Lammert and Allan, 1999). The size of the planning unit affects the total extent of additional protected areas needed to achieve representation targets. Because the extent of above target representation is lower, minimum sets built from smaller planning units are more efficient, requiring less area to achieve the same representation goals (Pressey and Logan, 1998; Rouget, 2003; Warman *et al.*, 2004) and therefore cheaper, and more likely to be implemented (Pressey and Logan, 1998). Smaller units can be amalgamated, enabling greater flexibility in the configuration and boundary placement of protected areas, and might be necessary for effective implementation in areas that are biologically heterogeneous or highly fragmented (Rouget, 2003). Conversely, larger units may result in minimum sets with higher contiguity and a lower mean perimeter-area ratio (Warman *et al.*, 2004) but usually need refinement for implementation, requiring additional study to accurately map locations of features to be protected by the designated planning units (Sundell, 1997; Cowling and Pressey, 2003).

Pressey and Logan (1998) list the number of units that can be considered in a reasonable time by the selection algorithms as a constraint on the size of units employed. However, because of rapid developments in computing power and disk storage this is now usually less of a constraint on the minimum size of planning units than is the availability of data on the occurrences of targeted biodiversity features. Spatial units also have an important role as a reporting framework where the appropriate size will be a compromise between a fine scale where it may be hard to appreciate the detail presented, and a very coarse scale at which over-generalised data become meaningless (Liston and Maher, 1997).

2.2.2. Type and configuration of spatial units

While the size of the spatial units is important, the type and configuration of spatial units also has the potential to influence conservation assessment outcomes. For example, public presentation of conservation assessment results by tenure parcels that can be individually identified are likely to be more sensitive than those presented via natural or arbitrary regular units (Pressey and Logan, 1998).

While design criteria for designated reserves (e.g. size, shape, connectivity) have long played a role (e.g. Margules and Stein, 1989; Lewis *et al.*, 1991), the addition of ecological and evolutionary processes to the targeted biodiversity features is a recent development in systematic conservation planning. The explicit consideration of spatial components associated with important ecological processes, such as drought refugia, reduces the chance that they are overlooked or incompletely covered by conservation plans (Pressey *et al.*, 2003). Although acquisition of lands for inclusion in an expanded reserve system may be easier where units are formed from tenure parcels (Cowling and Pressey, 2003), planning units that are formed from natural boundaries are more likely to facilitate the inclusion of these process components.

2.3. A spatial framework for rivers and streams

Clearly the choice of spatial framework does affect conservation planning and assessment. The framework scale influences the evaluation of conservation criteria and the distribution, number and total area of the selected sets while the types and configuration of planning units may affect their implementation and management. It is important, therefore, to consider carefully the reasons for using a particular spatial unit and planning domain (Warman *et al.*, 2004).

Not surprisingly, the conservation literature provides few recommendations to guide the choice of an appropriate spatial scale. No single scale is likely to be equally effective for all planning and assessment tasks or all biodiversity features of potential interest. A suitable framework should therefore, embrace a range of spatial scales. This would enable the scale of analysis to be varied to obtain the optimal resolution of pattern (Rahbek and Graves, 2000) or to match the

scale of the available data (Andelman and Willig, 2002). It should also assist the incorporation of ecological and evolutionary processes into the planning procedure and recognition of linkages between elements of the fluvial system, essential for managing conservation values.

In fact, river systems provide a natural spatial framework that satisfies these criteria. They encompass a nested hierarchy of units each operating within characteristic spatial and temporal scales (Frissell *et al.*, 1986; Naiman *et al.*, 1992) (Figure 2.1, Table 2.1). Micro-habitat features for example, vary across spatial scales of less than a metre and may persist for only a matter of days. Drainage basins on the other hand, are generally distinct geomorphic units representing patterns of freshwater connectivity that are stable over very long time frames. Drainage divides, the ridgelines that delimit drainage basins or catchments, generally define natural ecosystem boundaries (Stanford and Ward, 1992) and often will act as dispersal barriers for obligate freshwater species (Tait *et al.*, 2003). These hydrologic units form the logical basis for a hierarchical spatial framework for conservation assessment and planning. One level in the hierarchy will be used as the planning units; higher levels then provide the basis for assessing the context of these units (Bourgeron *et al.*, 2001d), reporting and protective management (Kingsford *et al.*, 2005).

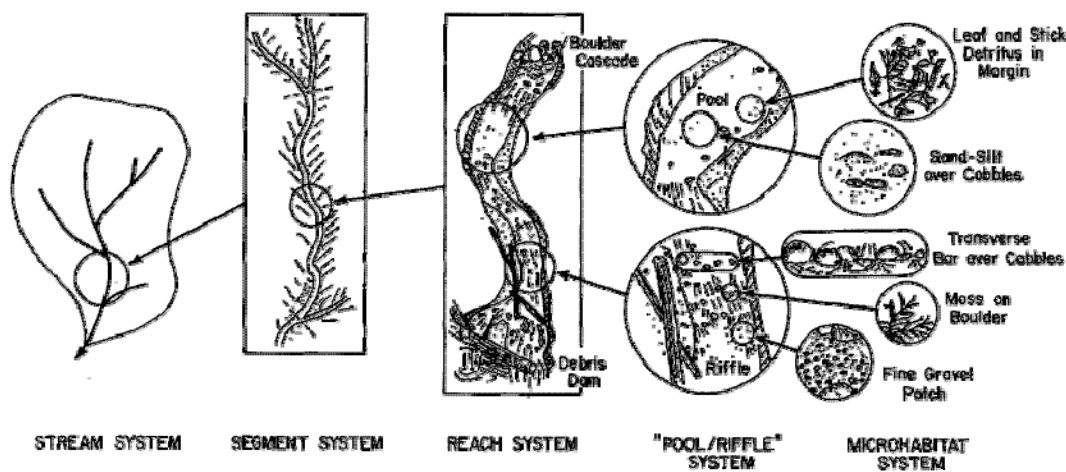


Figure 2.1. The hierarchical organization of a stream system (after Frissell *et al.*, 1986)

Table 2.1 A nested hierarchy of spatial units comprising a drainage basin. Adapted from Frissell *et al.* (1986) and Calvert *et al.* (2001).

Unit	Description	Time scale ₁ (years)	Linear scale ₁		Applicability at continental scale
			Small Streams (km)	Large Streams (km)	
Micro-habitat	Patch of similar flow velocity, substrate and cover	$10^{-1} - 10^0$	< 0.001	<0.1	Not possible
Habitat/ bedform	Areas of relatively homogeneous bed material, flow velocity and depth	$10^0 - 10^1$	0.001-0.01	0.1-10	Not possible
Reach	Length of river exhibiting relatively homogenous channel characteristics or a consistent pattern of repetitive/alternating characteristics	$10^1 - 10^2$	0.01 –0.1	10 –100	Not yet possible
Segment (link)	Portion of stream and its floodplain (including associated wetlands), bounded by tributary junctions, major waterfalls or lakes. The area of land draining to a segment or group of segments is a sub-catchment	$10^3 - 10^4$	0.1-1	100-1,000	Currently possible ²
Catchment	The area of land drained by a stream to a particular point (e.g. a tributary junction). May include internal sub-catchments	$10^4 - 10^5$	1-10	1,000-10,000	Currently possible ²
River basin	All of the catchment area that drains to a river mouth or terminal lake	$10^5 - 10^6$	> 10	> 10,000	Currently possible ²

1. Spatial and temporal scales are indicative only – actual values are appropriate to catchment size and mapping scale

2. At map scales of approximately 1:100,000 to 1:250,000

Not all of the potential scales are suitable as spatial units for continent-wide conservation planning (Table 2.1). Identification of spatial variability at the smallest scales (e.g. micro-habitat) is obviously impractical for large areas. Even for small area studies the large temporal instability make these unsuitable as conservation mapping units. River reaches are lengths of channel with uniform channel morphology or a consistent pattern of alternating channel morphologies (Calvert *et al.*, 2001). They are a common unit for field description for both fluvial geomorphologists and aquatic ecologists but usually require expert interpretation of low-level aerial photography or field survey to identify (Frissell *et al.*, 1986; Brierley and Fryirs, 2002). Consequently, reach differentiating criteria may vary considerably between operators and/or projects. Reaches may be wholly contained within a stream segment or overlap many segments. High resolution environmental data (e.g. terrain, geology) that may allow more automated and consistent reach delineation are not yet available continentally.

Catchments are readily delineated from a topographic map or, using drainage analysis software, from a Digital Elevation Model (DEM) (Section 3.2). Catchments have been widely adopted for natural resource planning and management and are often the basis upon which macro-regions are identified in freshwater bioregional frameworks (e.g. Abell *et al.*, 2000; Unmack, 2001; Tait *et al.*, 2003). Catchments may also be thought of as ecosystems (Stanford and Ward, 1992). However, the character of rivers may change greatly from their source to the ocean or a terminal lake or floodout (Calvert *et al.*, 2001) thus large catchment or basin scale analyses will generally be too coarse to provide useful levels of differentiation (Bennett *et al.*, 2002). Catchment scale conservation mapping units focus planning on large rivers, thus neglecting ecologically important small streams (Meyer and Wallace, 2000) and the distinctive aquatic fauna they may contain (Harding and Winterbourn, 1997; Baker *et al.*, 2004).

The most suitable spatial units to use as planning units for continental assessment and planning are stream segments i.e. that portion of a stream bounded by tributary junctions or major waterfalls. Segments, also known as stream links, have an associated sub-catchment, being that part of the catchment draining directly to the segment, and include associated riverine elements such as floodplain wetlands. They define natural discontinuities such as the abrupt changes associated with tributary inflows (Frissell *et al.*, 1986). Valley segments are identified as the hierarchical level most useful for landscape-scale analysis (Maxwell *et al.*, 1995). Likewise, Stanley and Boulton (2000) concluded that sub-catchments, as tractable and relatively homogenous spatial units, might be the most appropriate unit for river conservation and management. A hierarchical framework based at its lowest level on stream segments offers the opportunity to differentiate conservation values among both large and small streams. Segments can be assessed and managed within the context of the larger units (sub-catchments, catchments and drainage basins) in which they are nested; the larger units providing a more useful reporting

framework for national assessments. The nested units define and map distinct ecosystems at multiple spatial scales.

2.4. Conclusions

The conservation planning and assessment examples drawn from the literature illustrate the wide variety of spatial frameworks employed. The spatial framework is seen to be an important component in the planning and assessment process. The planning units provide the:

- geographic units to which data on the occurrence, frequency and extent of populations, species, assemblages or ecosystems are attributed for analysis
- selection or planning units which form the potential building blocks of a system of protected areas or priorities for protective management
- evaluation units to which conservation value is ascribed
- reporting units to present the results of conservation evaluation and progress towards conservation goals and the
- boundaries of potential areas for acquisition and/or application of protective management mechanisms

The spatial scale of the framework and the types and configuration of the planning units influence conservation planning outcomes. While these factors have been of research interest, the choice of spatial framework for many conservation planning programs has been dictated by pragmatic considerations. The boundaries of the assessment area or planning domain may reflect the responsibilities of the commissioning agency; rarely are they defined around ecological needs. A variety of planning units have been used to sub-divide the assessment area including administrative boundaries (tenure parcels, landuse), regular grids and natural regions (catchments, ecoregions). Almost exclusively, catchment based units have been the choice for river conservation planning exercises.

Catchments are, indeed, a sensible choice. They integrate many of the critical physical and biological processes that drive riverine ecosystems and thus may be appropriate units for description of many ecosystem patterns. However, a range of spatial scales is needed. The case is made for a hierarchical nested catchment framework built upon stream segments at its lowest level. The segment level represents a reasonable compromise between available scientific knowledge and tractability and the resolution necessary to distinguish conservation values and management actions within a catchment. This spatial resolution is also one that yields an appropriate balance between spatial and temporal predictability.

Chapter 3. Continental scale catchment frameworks

Chapter 2 describes the structure of a spatial framework that would underpin a national approach to systematic river and stream conservation planning in Australia. This framework is based on the natural nested hierarchical structure of river systems. However, existing catchment frameworks with continental coverage have serious shortcomings (Section 3.1) so a new system is required. This chapter focuses on how this might be developed (Section 3.2). Methods based on the analysis of a regular grid of elevation known as a DEM (Digital Elevation Model) enable consistent, automated delineation of stream and catchment networks. However, identifying where channels begin may not be easy (Section 3.2.1.1) and the algorithm chosen to represent surface flow paths may influence the outcomes (Section 3.2.1.2). Section 3.2.1.3 considers the procedures used to pre-process the DEM to improve drainage analysis outcomes. Repeatedly sub-dividing drainage areas into many smaller hydrologic units produces a nested hierarchical framework. Section 3.2.2 reviews the methods used to systematically delineate and uniquely identify these hydrologic units.

3.1. Existing catchment frameworks for Australian rivers and streams

The long standing and most widely adopted catchment framework for river management and planning in Australia is the Australian Water Resources Council (AWRC) drainage basins and divisions (Australian Water Resources Council, 1976; AUSLIG, 1997; Geoscience Australia, 2003a) (Figure 3.1). The 245 drainage basins so identified are based on the catchments of the major river systems, but also include the catchment areas of neighbouring small coastal or inland drainage systems. These basins are aggregated according to geographic proximity or shared discharge points to form 12 drainage divisions. However, the AWRC river basins confuse a number of spatial scales. They include topographically defined basins (e.g. the Fitzroy River in Queensland), the catchments of major rivers (e.g. of the Murray-Darling Basin) and sub-catchments (e.g. the lower Avon River in Western Australia). The valley segment scale is not recognised. Moreover, a number of the basin boundaries delineate convenient administrative units, rather than true catchment divides. The boundary between the Paroo and Warrego River cuts through a distributary that joins the rivers (Kingsford *et al.*, 2001) while the Murray River forms the divide between basins within the Murray Darling Basin. The boundaries of the Western Plateau drainage division ignore major palaeodrainage patterns that shaped the landscape of central and western Australia (Hutchinson and Dowling, 1991). Although useful as

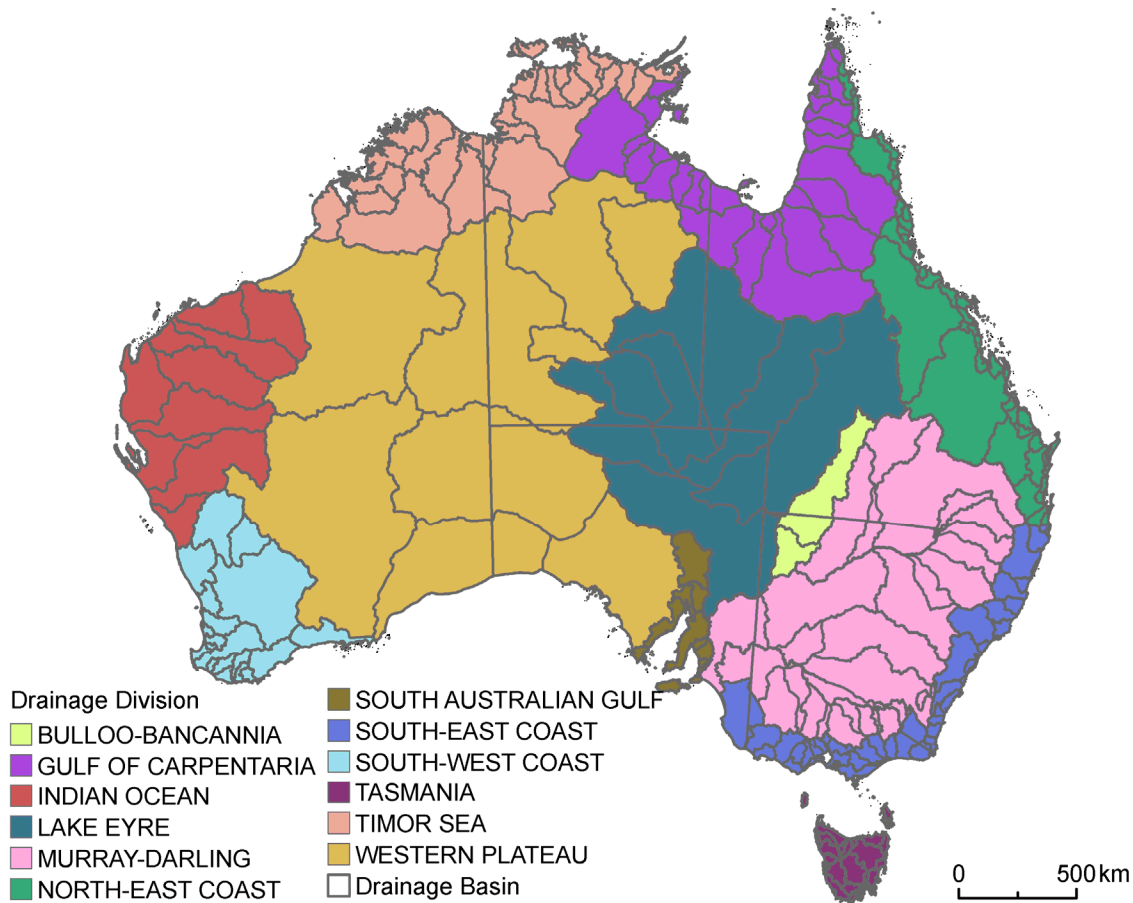


Figure 3.1 The Australian Water Resources Council (AWRC) Drainage Divisions and Basins (AUSLIG, 1997).

a reporting framework, the lack of adherence to topographically defined hydrological boundaries, errors in boundary location and coarse resolution limit their conservation planning use.

A set of nested sub-catchments were delineated by Hutchinson *et al.* (2000) to support the activities of the National Land and Water Resources Audit (NLWRA). The levels in the hierarchical sub-division were defined by application of increasingly larger contributing area thresholds (2.5, 25, 50 and 500 km²) using the standard methods for drainage analysis of a DEM (Section 3.2.1). Threshold values were dictated by the requirements of NLWRA projects. These sub-catchments thus provide a much finer sub-division of catchments than the AWRC basins and are closer to providing the desired multi-scale framework. However, two serious problems limit their utility. Firstly, they fail to recognize distributary drainage structures or the anabranching streams that link a number of catchments at high flow levels thus prejudicing the analysis of biogeographic patterns and processes. Secondly, the patterns of drainage density derived by application of uniform area thresholds are inconsistent with observed patterns in some areas, especially those of low relief or extreme aridity (Figure 3.2).

Australia is also covered by a number of global drainage datasets (e.g. Renssen and Knoop, 2000; Vörösmarty *et al.*, 2000; Döll and Lehner, 2002). These datasets were developed from

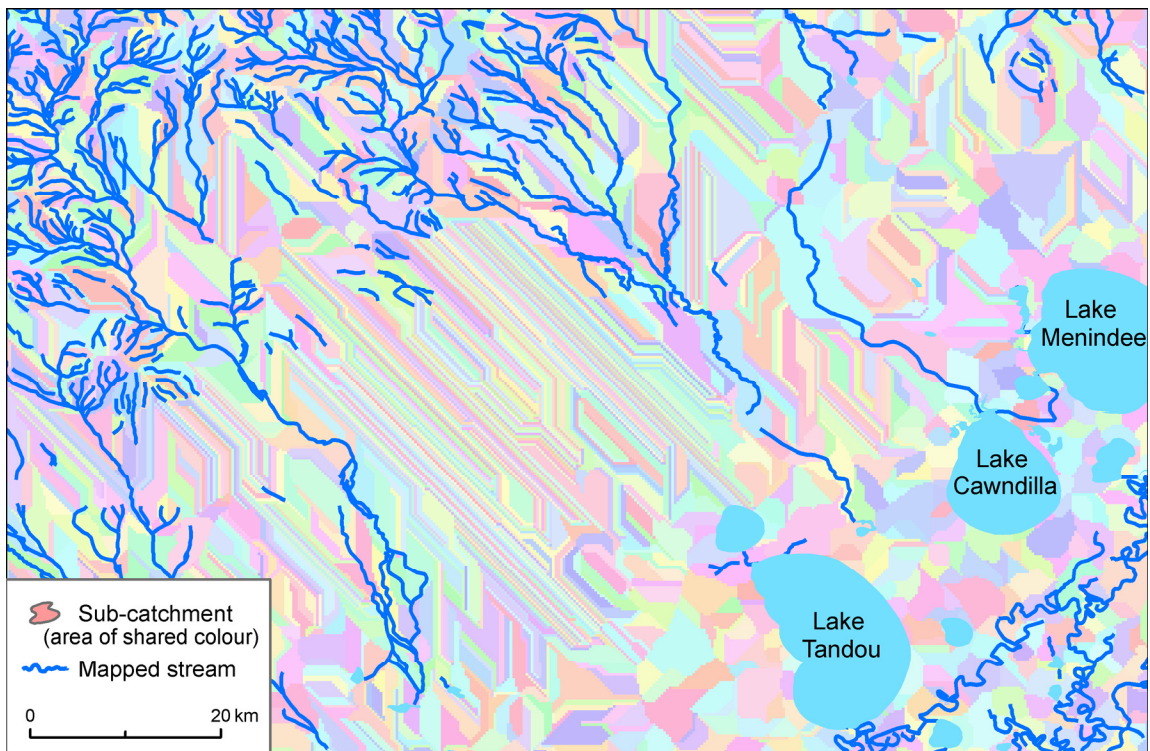


Figure 3.2. Sub-catchments from the nested catchments layer defined for the NLWRA (Hutchinson, Stein *et al.*, 2000) using a contributing area threshold of 2.5km^2 . Shown is an area near Menindee Lakes on the Darling River in western New South Wales. The linear sub-catchments evident in the centre of the figure are artefacts often seen in areas of low relief where surface drainage patterns are poorly defined by continental scale DEM's. The imposition of a uniform area threshold also produces a level of catchment sub-division implying a much higher drainage density than is observed.

DEMs at spatial scales of $30'$ to 1° spatial resolution or coarser as a basis for continental and global scale modelling of water and sediment transport. Döll and Lehner (2002) reported concerns about the quality of some of these global datasets. None represents flow bifurcations and only larger rivers are differentiated at these scales. Vörösmarty *et al.* (2000), for example, found first order streams in their global river network, STN-30p (Simulated Topological Network), were roughly equivalent to an order five or six river derived from 1:62,500 scale maps. Small drainage basins (less than $25,000\text{km}^2$) could not be characterized with confidence. The HYDRO1k dataset (U.S. Geological Survey, 2001), regarded as the most useful of the global coverages (Döll and Lehner, 2002), includes streams, drainage basins and ancillary layers (e.g. slope, aspect, contributing area) derived from the USGS' 30 arc-second DEM of the world (GTOPO30). HYDRO1k delivers a nested hierarchical catchment framework by successively sub-dividing drainage basins according to the Pfafstetter scheme (Verdin and Verdin, 1999). However, as of March, 2005, coverage for Australia was still not available and other problems, such as streams crossing drainage divides are reported (Revenga *et al.*, 2005). Work is underway to replace the HYDRO1k products with higher resolution derivatives of the new SRTM shuttle DEM (Farr and Kobrick, 2000), although the algorithms used to derive these products (Curkendall *et al.*, 2003) ignore distributary drainage systems and may have difficulties with endorehic drainage.

In summary, all existing stream and catchment databases with Australia-wide coverage have serious shortcomings. To satisfy the requirements of the proposed spatial framework a new database is needed. A variety of methods is available to undertake this development.

3.2. Delineating a new catchment framework

Traditionally, paper maps have been the source of spatial information for river management and planning, at map scales of 1:25,000 or finer for detailed stream survey and mapping (e.g. Brierley and Fryirs, 2002). Drainage divides are identified by manually analysing contour lines, a difficult, time-consuming task that may be subject to operator errors and inconsistencies (Maidment, 2001). Paper maps may be a suitable basis for small catchment scale studies, but automated methods are needed to support consistent and rigorous continental scale conservation analysis. While the streamline information portrayed on topographic mapsheets is now compiled in digital format suitable for GIS analysis (e.g. the GEODATA TOPO 250K database AUSLIG, 1992b; Geoscience Australia, 2003c), the vector formats employed present difficulties for common drainage analysis tasks such as stream and catchment delineation, flow routing and computation of catchment areas and network order. For example, there is often significant variability in mapped drainage density between map sheet tiles due to variable cartographic interpretation (Figure 3.3) – thus streamlines appear to initiate or terminate abruptly at mapsheet borders. The complex drainage patterns that topographic mapsheets depict, including unconnected or extensively anabranching or anastomosed channel systems and numerous abandoned channels (e.g. billabongs), make tasks such as flow routing particularly difficult. As a result, drainage analysis methods based on a regular grid of elevation known as a Digital Elevation Model (DEM), are more commonly used (Moore *et al.*, 1991; Rieger, 1998; Turcotte *et al.*, 2001; Tarboton, 2004). Triangular Irregular Networks (TIN's), irregularly spaced nodes that sample surface specific points, such as ridges, peaks and breaks in slope and contour-based data structures that store x,y co-ordinate pairs along each contour of specified elevation, are also employed to represent topography (Moore *et al.*, 1991). Grids are generally preferred, however, because they are simpler, readily compatible with other types of data, require less data storage and constitute an uncomplicated target for model programming (Holmgren, 1994; Bertolo, 2000).

3.2.1. Drainage analysis of a DEM

The most widely adopted approach follows the method of Jenson and Domingue (1988) (e.g. Hutchinson and Dowling, 1991; Jenson, 1991; ESRI, 1996; Norris *et al.*, 2001). It depends on the derivation of a grid of flow directions (O'Callaghan and Mark, 1984) that records the direction of steepest descent for each individual cell to one of its neighbours (Figure 3.4). In

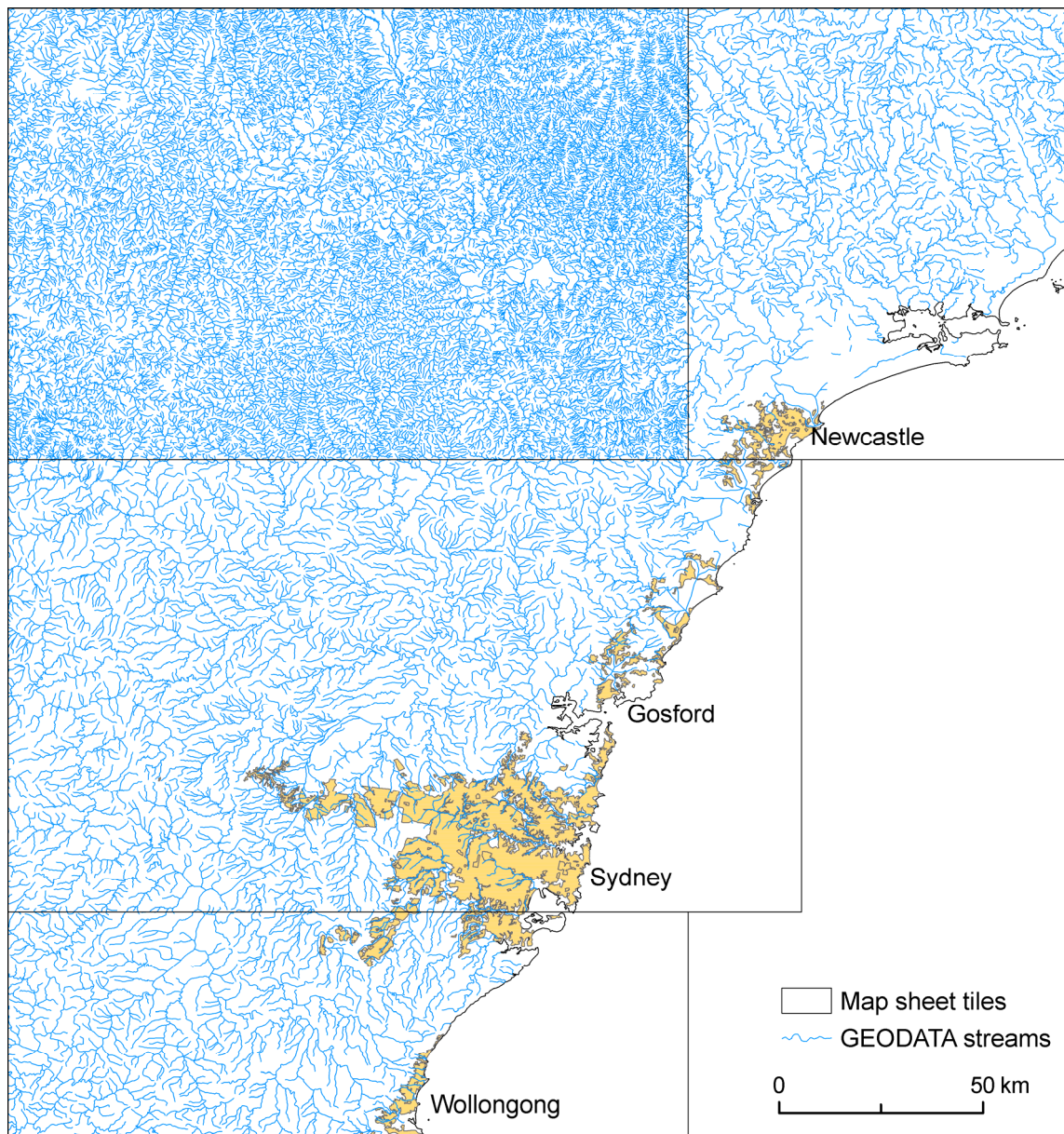


Figure 3.3. Streamlines from the GEODATA TOPO 250K database version 1 (AUSLIG, 1992b) for the area around Sydney. Note the dramatic differences in drainage density that coincide with the 1:250,000 mapsheet tiles. Version 2 of the GEODATA TOPO 250K database (Geoscience Australia, 2003c) addresses this issue by removing streamlines from excessively dense areas but does not appear to extend streamlines where they abruptly end at map tile borders.

this, the D8 method, flow path configurations are restricted because flow can only occur in one cardinal or diagonal direction. The upstream flow paths delineate the contributing area for each cell and therefore can be used to map the catchment above a set of grid cells or “seeds”. The seeds, for example, all confluences of a stream network (Jenson, 1991), represent the outflow points of the desired catchments. Gridded representations of mapped streamlines have been used as catchment seeds (Stein *et al.*, 2002), however, where valleys defined by the DEM do not correspond precisely with the vector streamlines, errors and artifacts may result (Norris *et al.*, 2001) and special procedures may be required to correctly represent network topology in flow routing applications (Stein *et al.*, 2002).

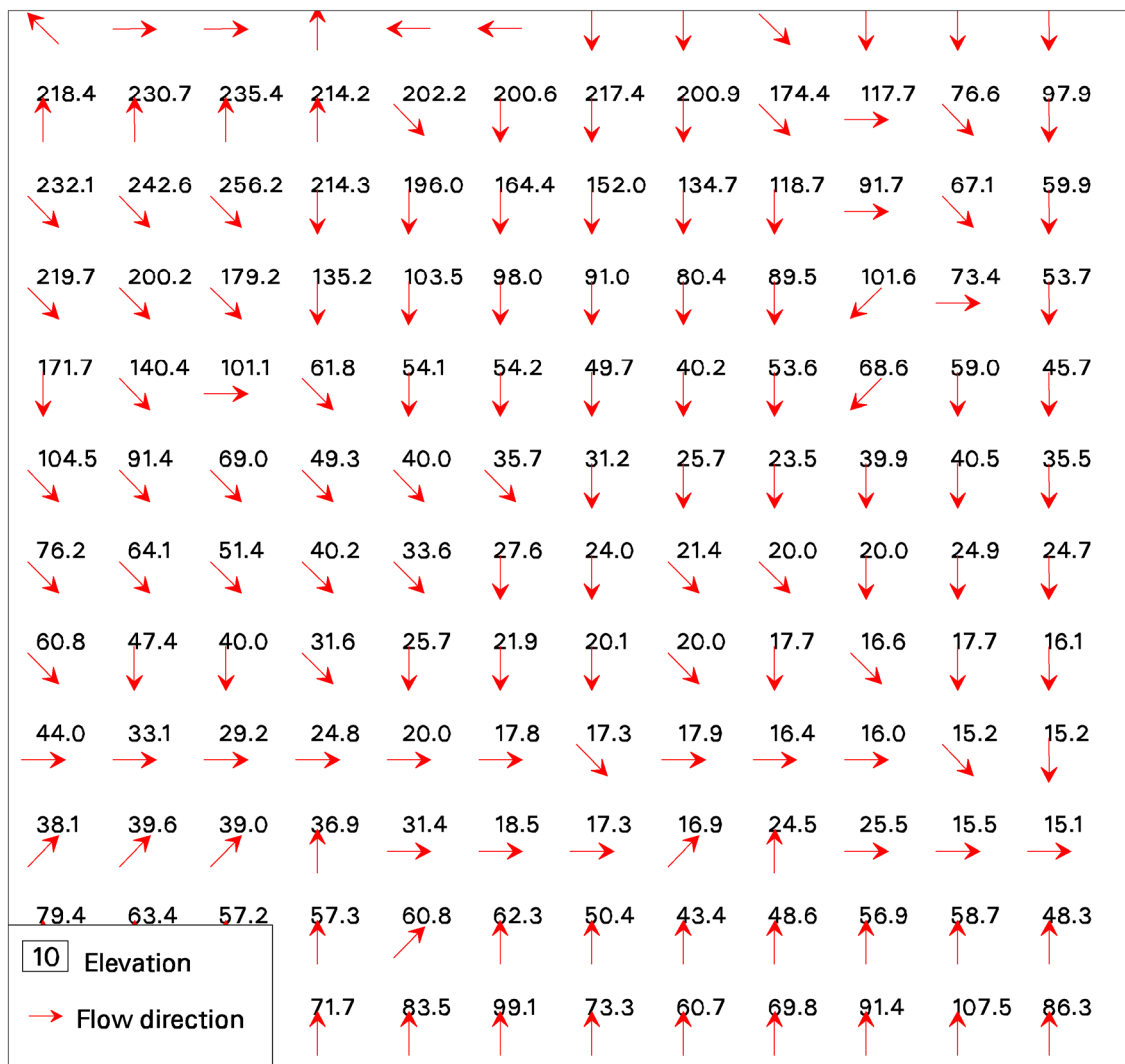


Figure 3.4. The D8 method of determining flow direction. The direction of flow from one cell to the next is that in the direction of steepest slope.

3.2.1.1. Delineating a stream network

A variety of methods has been used to automatically delineate a stream network from a DEM using characteristics of the local landscape morphology, such as terrain curvature (e.g. Tarboton and Ames, 2001). Most often however, stream networks are delineated by application of a critical support area threshold that defines the minimum contributing area required to initiate a channel (Jenson, 1991; Montgomery and Foufoula-Georgiou, 1993; Verdin and Verdin, 1999; Hutchinson, Stein *et al.*, 2000) (Figure 3.5). The threshold is assumed to represent the level of runoff concentration at which fluvial processes dominate over hillslope processes (O'Callaghan and Mark, 1984). Grid cells with an upstream contributing area greater than the specified threshold are considered to belong to the channel network. The choice of threshold value determines the density of the resulting drainage network; increasing the threshold contributing area leads to an exponential decrease in the channel network extent and the drainage density (Thieken *et al.*, 1999). In practice, threshold values have often been chosen to suit the requirements of the application (Hutchinson, Stein *et al.*, 2000) or to delineate channel networks

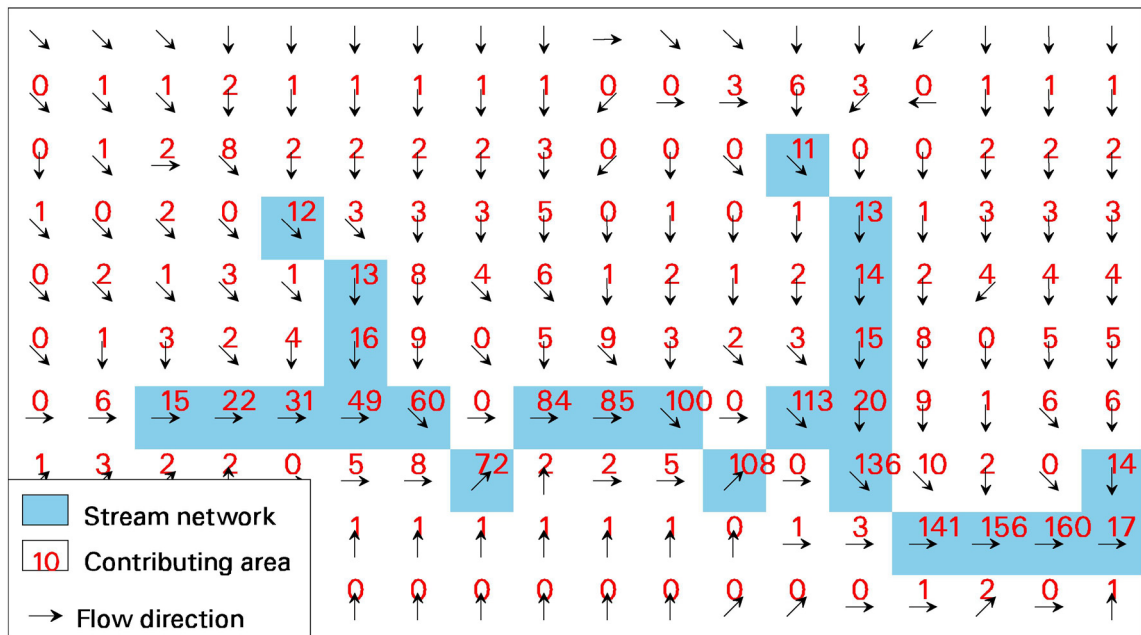


Figure 3.5. Delineating a stream network by application of a contributing area threshold . Cells with contributing areas above the designated threshold are considered to belong to the stream network. Contributing areas are computed by accumulating the area of all upstream cells according to the flow direction.

that most closely resembled the streamlines shown on topographic maps (Montgomery and Foufoula-Georgiou, 1993). However, these streamlines reflect subjective cartographic interpretation and generalization and may underestimate the actual channelised flow network (Montgomery and Foufoula-Georgiou, 1993; Gandolfi and Bischetti, 1997; Knighton, 1998; Hansen, 2001).

Tarboton and his colleagues (1991; 1992) proposed quantitative means of identifying appropriate threshold values, invoking the observed scaling laws of channel networks (constant drop property and power law scaling of slope with area). They suggested the contributing area threshold could be identified from the breakpoint in the log-log plot of link slope against contributing area, assumed to differentiate the change from diffusive hillslope to fluvial channel processes. Montgomery and Foufoula-Georgiou (1993) subsequently pointed out a number of problems with Tarboton *et al.*'s implementation suggesting an alternative formulation based on local, rather than link (segment) slopes. They also noted that the point of inflexion generated threshold values that implied anomalously large hillslope lengths and that using the point of reversal in the direction of the slope-area relationship was more consistent with theory.

The TAUDDEM terrain analysis software package (Tarboton, 2004) implements the constant drop scaling procedure. The program identifies the smallest contributing area value that delineates a stream network consistent with the "constant drop law" i.e. a network where the mean stream drop of first order streams is not significantly different from the mean drop of higher order streams. Stream drop is the difference in elevation between the beginning and end

of Strahler streams, where a Strahler stream is an entire set of sequential stream segments of the same stream order.

However, no single threshold value is likely to be appropriate over large areas with heterogeneous terrain types in which different geomorphic processes dominate (Soille *et al.*, 2003). Applying a uniform contributing area threshold to delineate a stream network may overcome some of the limitations of topographic mapping attributed to cartographic interpretation, but is not adaptive to the spatial variability in drainage density. Accounting for the role of the local slope in the channel initiation process provides a mechanism for spatially variable drainage density (Tarboton and Ames, 2001). The observed inverse relationship between gradient and catchment area for channel heads (Montgomery and Dietrich, 1989; Montgomery and Foufoula-Georgiou, 1993), yields a slope dependent channelization threshold of the form

$$aS^{\alpha} > C$$

where a is specific catchment area and S is slope, α is usually equal to 2 and C is a proportionality constant that depends on both ground surface (soil properties and vegetation cover) and climate (Montgomery and Foufoula-Georgiou, 1993). This model produces higher drainage densities where slopes are steeper, as observed for natural landscapes, leading Montgomery and Foufoula-Georgiou (1993) to suggest that it is both theoretically and empirically a more suitable model than the constant contributing area threshold for defining the extent of channel networks, except where bedrock properties control channel head locations. As the value of aS^{α} can decrease downstream with declining slope this model is only necessarily valid at channel heads (Lindsay 2003).

Again, however, identifying a suitable threshold may be difficult. A uniform threshold will not be appropriate across large areas. Field observations reveal considerable noise in the relationship between slope and catchment area at channel heads (e.g. Menduni *et al.*, 2002) and it appears that the gradient of this relationship may vary across drainage basins (Kirkby *et al.*, 2003; Moeyersons, 2003). Tarboton and Ames (2001) also report that applying the slope-dependent area threshold produces “feathering” of the drainage network in steeper areas while underestimating the extent of the drainage network in shallower valleys. Montgomery and Foufoula-Georgiou (1993) attribute feathering to the use of an inappropriate value for the proportionality constant C , suggesting the Tarboton and Ames conclusion could be the result of applying a uniform threshold value across a geomorphologically heterogeneous area.

Istanbulluoglu *et al.* (2002) characterize the observed variability in the channel initiation threshold by treating C as a random variable. Their model derives a spatially variable probability of channel initiation that depends on slope, specific catchment area, and the probability distributions of median grain size, surface roughness, and excess rainfall rate. In this

model, the channel initiation zone on the slope-area plot varies depending on model input values that characterize climate and landcover variability.

Vogt, Colombo and Bertolo (2003) recognized a broad range of environmental controls on drainage density needed to be considered in order to derive continentally applicable contributing area thresholds. They first identified landscape types delineated according to values of a drainage density index. The index was derived as a function of environmental factors (climate, vegetation cover, relief, soils and lithology) that were believed to exert a strong control on channel initiation and therefore, the development of the stream network. Type specific contributing area thresholds were then determined by identifying the breakpoint on plots of the slope-area relationship for each landscape type. Vogt and his colleagues applied the method in Italy yielding threshold values of 3, 10, 50, 100 and 700 km² respectively for each of the five landscape types. While they found the derived patterns of spatial variability of drainage density were reasonable, the threshold values are significantly greater than those reported by other workers, suggesting that they probably underestimated the extent of the true channel network. Wasson (1998), for example, notes that the upstream catchment areas of headwater channels are generally of 1 ha or less. Similarly, Montgomery and Foufoula-Georgiou (1993) find contributing areas of considerably less than 1 km² at field mapped channel heads.

These discrepancies highlight the difficulties that confront automated delineation of channel networks over very large and heterogeneous areas. There is a significant mismatch between the scale of the channel initiation processes that these methods seek to model and that of the available information on the controlling factors. Vogt, Colombo and Bertolo (2003), for example, used soil transmissivity classes derived from soil texture information provided on the 1:1,000,000 scale European Soil Map to describe the influence of soil properties on drainage density, but at this scale mapping units include considerable variability in soil parameters (Jamagne *et al.*, 2001). Hillslope lengths, directly correlated with drainage densities are typically in the range of tens to hundreds of metres (T. Dunne unpublished data cited in Montgomery and Foufoula-Georgiou, 1993; Gallant, 2001), in many instances smaller than the size of an individual grid cell in the 250m resolution DEM used by Vogt *et al.* in their Italian study. The accuracy of slopes derived from the DEM is a critical limitation on the application of slope-dependent thresholds (Roth, LaBarbera *et al.*, 1996). Derived slopes are particularly sensitive to the DEM resolution. As the grid cell size increases short, steep slopes are increasingly likely to be missed (Wilson *et al.*, 2000; Gallant, 2001; Clarke and Burnett, 2003). Even high resolution (1m) DEM's typically underestimate true slopes (Warren *et al.*, 2004). Indeed, Montgomery and Foufoula-Georgiou (1993) concluded that the extent of the channel network could not be directly determined from the drainage area-slope relations extracted from a DEM and that field data were necessary to identify appropriate parameters. Miller (2002) also

reasoned that it was not feasible to accurately locate channel heads from a DEM and therefore the goal should be to select threshold values that reproduce realistic drainage densities.

3.2.1.2. Flow direction

Computation of contributing area is also sensitive to DEM resolution but, more importantly, to the choice of flow direction algorithm. Contributing areas are calculated by accumulating the area of all grid cells upstream according to the flow paths defined in a flow direction grid (Figure 3.5). The widely adopted D8 method (O'Callaghan and Mark, 1984) (Figure 3.4) allows flow from a cell to only one of its eight neighbours and cannot model flow dispersion. It tends to produce flow in parallel lines, biased towards aspects in multiples of 45°, resulting in significant errors in contributing area values (Costa-Cabral and Burges, 1994; Gallant and Wilson, 2000; Pan *et al.*, 2004). Multiple flow direction algorithms (Freeman, 1991; Quinn *et al.*, 1991; Holmgren, 1994; Tarboton, 1997) overcome this limitation by distributing flow from a grid cell amongst its lower-elevation neighbours according to some specified rule. The FD8 algorithm (Quinn *et al.*, 1991), for example, passes flow to all lower neighbours proportionally on a slope-weighted basis. Multiple flow direction methods give more realistic distributions of contributing areas in upslope areas than single flow direction methods such as D8 (Tarboton, 1997; Gallant and Wilson, 2000; Wilson *et al.*, 2000). They produce fewer artefacts and are, therefore, generally recommended for modelling the distribution of contributing area across hillslopes (Gallant and Wilson, 2000; Pan *et al.*, 2004).

However, while the multiple flow direction algorithms may give a more realistic pattern of accumulating area on the hillslope portion of the catchment, they may also create undesirable dispersion of flow in valleys where convergent flow would be better represented by the D8 method (Quinn *et al.*, 1991). Thus, implementations of multiple flow direction algorithms generally switch to D8 when either the computed contributing area reaches a user-specified threshold (Gallant and Wilson, 1996) or convergent flow is detected (Freeman, 1991). To avoid the abrupt changes that may occur at the transition to channelized flow (Gallant and Wilson, 2000), spatially varying flow apportioning methods have been introduced to control the amount of flow divergence depending on the relative position within the landscape (Lindsay, 2003; Kim and Lee, 2004).

Flow direction is critical to a number of other drainage analysis tasks such as delineating catchment areas and stream network analysis. General purpose Geographic Information System (GIS) software, such as ArcInfo (ESRI, 2004), implement only the basic D8 algorithm. However, dedicated software packages for terrain and hydrological analysis, for example, TAPES-G (Gallant and Wilson, 1996), TAS (Terrain Analysis Software) (Lindsay, 2005), TauDEM (Terrain Analysis Using Digital Elevation Models) (Tarboton, 2004), RiverTools (Rivix LLC, 2004) and CatchmentSIM (Ryan and Boyd, 2003), provide at least one other flow

direction algorithm. Nevertheless, tasks that require unique, non-diverging flow paths, such as computing up or downstream flowpath lengths, still rely on the steepest descent or D8 algorithm (Lindsay, 2005). The D8 algorithm is also used to delineate catchment boundaries (Gallant and Wilson, 2000). The numerous outflow paths that can result from the application of multiple flow direction algorithms create “diffuse and overlapping” catchment boundaries (Jones, 2002). In contrast, because the D8 method allocates each grid cell to only one catchment, data storage, analysis and visualization are simplified.

3.2.1.3. DEM preparation

Drainage analysis methods rely upon the DEM accurately representing surface drainage pathways. However, inadequacies in many DEM's require significant pre-processing or filtering that may include the filling of spurious sinks and the imposition of the mapped streams onto the DEM (“burning in”).

Sinks or pits in a DEM are grid cells or groups of cells with no neighbouring cells of lower elevation. The drainage direction of these grid cells is therefore undefined and they do not contribute flow to any other cells, terminating flow paths. Their presence in DEMs is usually attributed to data errors, shortcomings of interpolation procedures, the limited horizontal and vertical resolution of the DEM or as a consequence of the type of surface representation (Jenson and Domingue, 1988; Martz and Garbrecht, 1998; Rieger, 1998; Hutchinson and Gallant, 1999). Spurious sinks frequently pose serious problems for hydrological applications (Hutchinson and Gallant, 2000). Many procedures have been proposed to eliminate them and so improve drainage analysis outcomes.

One widely used corrective measure is that attributed to Jenson and Domingue (1988). The elevation of sink cells is raised to the height of the lowest outlet through which water will flow, i.e. the pour point, and flow directions recomputed from the ‘filled’ DEM. A threshold value for the depth of the depression from the surrounding surface elevation may be used to restrict the fill to more localised sinks (e.g. Figurski and Maidment, 2001).

A major disadvantage of this approach is that all information about terrain shape within the depression is lost. Additional processing is then required to assign flow directions, for example using topological relationships (Jenson and Domingue, 1988), within the resulting flat areas. This approach works on the unrealistic assumption that all depressions result from an underestimation of DEM elevations (Martz and Garbrecht, 1998). Rieger (1998) instead suggests elevation points that are too high at the outlets as the reason for many depressions. Accordingly, he describes a method to lower the flow path, starting from the outlet. Other workers (Martz and Garbrecht, 1998; Jones, 2002; Ryan and Boyd, 2003; Soille, 2004) have also proposed similar “breaching” or more radical “carving” procedures. These approaches to sink removal alter fewer cells and avoid the creation of large flat areas. Soille (2004) found,

however, that a hybrid approach combining pit filling and carving minimized the differences between the original and modified DEM.

Carving procedures can be directly extended to adaptive drainage enforcement (Soille *et al.*, 2003). Drainage enforcement, stream burning (Saunders, 1999) and surface reconditioning (Hellweger, 1997) are all procedures designed to correct the deviations between stream networks delineated from a DEM and the bluelines on maps, usually due to inadequate DEM vertical resolution in areas of low relief (Saunders, 1999). They involve the imposition of vector streamlines onto a DEM to force the DEM flow pathways to conform to the vector representation, either by artificially lowering the grid cells underlying a stream channel or incrementing the non-stream cells (Hellweger, 1997; Saunders, 1999; Ryan and Boyd, 2003). In the case of stream burning, a fixed elevation differential is universally applied. While the revised flow pathways derived from the “burned-in” DEM accord better with the mapped streamlines, new errors and artefacts may be introduced. Reported problems include the introduction of artificial parallel streams into the derived channel network (Hellweger, 1997), distortion of catchment boundaries and meandering stream paths (Saunders, 1999) and substantial alteration of slopes around the streamline (Gallant and Wilson, 2000). Hellweger (1997) and Turcotte *et al.* (2001) introduce mechanisms for adjusting the elevation differential according to the distance from a mapped streamline, overcoming some of the shortcomings of the original stream burning procedures. The method of Turcotte and colleagues also incorporates lakes so that they may be differentiated from flat areas in the DEM. All of these methods however, require extensive editing of vector format data before application (Saunders, 1999; Turcotte *et al.*, 2001).

Automatic DEM filtering or “reconditioning” procedures are widely adopted to overcome the problems of spurious sinks or poorly defined flow pathways in areas of low relief (e.g. Renssen and Knoop, 2000; Vogt, Colombo, Paracchini *et al.*, 2003). However, such methods can also alter well-defined surface features in a DEM (Hutchinson, 1989). Genuine sinks do occur, for example, in areas of aeolian deposition with dune and swale topography or ephemeral lakes and claypans (Mark, 1988). Realistic analysis of surface drainage pathways must represent these geomorphic environments that are widespread in the arid interior of Australia.

DEM pre-processing could be largely avoided by careful attention to source data errors and inadequacies (Hutchinson and Dowling, 1991; Hutchinson *et al.*, 2001) and the use of intelligent terrain interpolation routines for DEM development. The ANUDEM program (Hutchinson, 1989, 2004a) is one such interpolation routine. By addressing many of the shortcomings of DEM's cited above, it averts the need for significant DEM pre-processing. ANUDEM produces a DEM from irregularly spaced elevation point or contour data and directed streamline vectors. The program automatically removes spurious sinks by imposing a global drainage condition on the fitted elevation grid. It interpolates the elevation data by

minimizing a suitably weak roughness penalty and imposing constraints directly from the streamline data to ensure a connected drainage structure. Program diagnostics (non-data point sinks, large residuals and drainage enforcements) provide an efficient mechanism for identification of source data errors in large data sets. The built-in drainage enforcement accurately represents drainage structure in low relief areas (Hutchinson and Gallant, 2000) removing the subsequent need for stream burning or other DEM “reconditioning” procedures.

3.2.2. Hierarchical catchment reference systems

Automatically delineating a stream network from a DEM does present a number of challenges, but delineating catchments is relatively straightforward as they are directly associated with streams via the derived flow pathways (Bertolo, 2000). Stream networks automatically delineate the stream segment level of a drainage basin sub-division. The grid cells draining directly to the segment form the segment sub-catchment. However, it is less obvious how higher levels in a hierarchical catchment system should be defined. Accordingly, many different systems have been employed (Table 3.1).

Several examples were introduced in Section 3.1. The AWRC system used regional affiliations to group catchments around the major river systems. It applied no consistent scheme to sub-divide drainage basins. The Hydrological Units System (HUC) of the United States also groups adjacent catchments, but the nested hierarchy of 21 regions, 222 subregions, 352 accounting units, and 2,149 cataloguing units is principally organized to achieve similar size areas at each level (FGDC, 2002). It is being updated and extended to include 5th and 6th level sub-divisions, mapped at a scale of 1:24,000 (Berelson *et al.*, 2004). Like AWRC, many of the HUC units are not true topographic catchments (Griffith *et al.*, 1999) and although guidelines and interagency standards are issued to encourage consistency, different rules may be used to sub-divide HUC units into a variable number of smaller units (FGDC, 2002).

In contrast, the levels in the nested catchments of Hutchinson, Stein *et al.* (2000) are formed consistently by applying successively larger catchment area thresholds to a grid of contributing area. The outlets of each sub-catchment unit are thus located at the confluences of tributaries with catchment area greater than the required threshold value. The nested catchments contain tens or even hundreds of thousands of units at each level, each of which was uniquely identified with a sequential number. While this numbering scheme was easily applied, it conveys little useful information.

Desirable features of a coding system include the ability to code membership of higher levels and topological relations (tributary and mainstem, up/downstream linkages) and enable division into a finite number of topographically defined catchment units at any hierarchical level. It will

Table 3.1 Systems for delineation and/or codification of topographically defined hydrologic units at national scales

System	Description	Automated delineation/ coding?	Information conveyed by unit coding scheme		
			Membership of higher levels	Tributary/ main stem relations	Up/down stream relations
Pfafstetter (Verdin and Verdin, 1999)	Drainage basins successively divided into tributary basins and mainstem inter-basins on the basis of catchment area and drainage network topology.	Y	Y	Y	Y
USGS Hydrologic Unit System (HUC) (FGDC, 2002)	Nested hydrologic units. Criteria for the numbers of units and area guide division into smaller units. Downstream boundary decisions made using local knowledge. Inter-agency standards and guidelines facilitate consistency.	N	Y	N	Y
U.S. EPA Reach File 1 (RF1) (U.S. Environmental Protection Agency Office of Science and Technology, 1998)	Stream reach indexing system using a unique reach number for stream reaches and relative position (latitude/longitude), pro-rated against the full computed reach length.	Y	N	N	Y
Catchment area thresholds (e.g. Hutchinson, Stein <i>et al.</i> , 2000)	Unit downstream boundaries located at confluences where the tributary upstream catchment area exceeds specified threshold.	Y	Y	N	N
AWRC Drainage basins and Divisions (AUSLIG, 1997; Geoscience Australia, 2003a)	Based on catchment area of largest rivers, adjacent small catchments merged. Basins numbered sequentially within drainage divisions. First digit indicates drainage division.	N	Y	N	N
Brooks, 2003	Hydrological coding system similar to Pfafstetter using base 64 numbering system requiring less sophisticated operators for up/downstream queries.	Y	N	Y	Y

have a numbering system that is self-replicating at finer scales to accommodate further sub-divisions as required. The system should also be easy to implement and understand.

The Pfafstetter system stands out as an effective coding scheme (Table 3.1). The system, described by Verdin and Verdin (1999) based on concepts developed by Otto Pfafstetter, a Brazilian engineer, delineates a hierarchical, spatially nested catchment reference system. It defines and numerically codes topographically defined sub-basin units guided by the topology of the drainage network and the size of the drainage area. The system is efficient, using only the digits 0 to 9 for each level of sub-division, significantly fewer digits than other coding schemes (Verdin and Verdin, 1999). Consequently, continents can be sub-divided into a large number of hydrological units with relatively short and manageable identification numbers. The codes convey useful information about topological relations in a catchment. For example, all sections of a river network either up or downstream of any feature of interest are readily inferred from the codes using simple algebraic queries.

The Pfafstetter scheme was adopted for the European-wide stream and catchment database (Vogt, Colombo, Paracchini *et al.*, 2003) following a comprehensive review of alternative coding schemes (Britton, 2002). Phillips *et al.* (2002) used the Pfafstetter system to delineate nested catchment units for a trial of a new method of conservation assessment for the Burnett River catchment in Queensland and it is the basis of the global HYDRO1k nested catchment coverage (U.S. Geological Survey, 2001). German water authorities use a similar scheme, the Länderarbeitsgemeinschaft Wasser (LAWA) system (Britton, 2002). LAWA reverses the direction of the Pfafstetter numbering, instead numbering sub-basins from the headwaters to the river mouth. The ERICA coding method is another system based on Pfafstetter (Bertolo, 2000). It extends the code for each level to two digits so catering for a larger number of sub-catchment divisions (up to 99 sub-catchments and 49 tributary basins) but the resulting codes are harder to interpret and less economic in the use of digits (Britton, 2002).

Brooks (2003) proposed an alternative coding scheme built on a base 64 numbering system that enables rapid up or downstream tracing with less sophisticated query operators than those required for the Pfafstetter scheme. However, this is not a serious limitation of Pfafstetter as the required operators (the ability to subset strings and the modulus operator) are readily available in commonly employed tools such as the spreadsheet program, Microsoft Excel and the Arc/Info GIS package. Furthermore, the Brooks scheme is more difficult to explain and interpret and less efficient in the number of digits employed.

Britton (2002) reviewed a number of other catchment sub-division and coding schemes in use in Europe, but found none were able to match the Pfafstetter scheme on important criteria of ease of implementation and interpretation, global applicability, economy of digits and conveyance of useful information within the code.

3.3. Conclusions

The existing catchment frameworks for Australian rivers and streams have serious deficiencies. None fully meets the requirements for continent-wide systematic conservation planning. A new stream and catchment spatial framework is therefore required.

Automated methods of drainage analysis delineate stream networks and catchment boundaries from a DEM, providing a number of advantages over traditional, manual procedures. Applicable across large areas in a consistent, repeatable and timely manner, they offer the most suitable approach for the development of a new Australian stream and catchment framework. The drainage analysis methods have been widely applied in catchments with connected dendritic drainage patterns such as those found in many upland settings in Australia. However, there appears to be no continentally applicable solution to the difficulties posed by the distributed and uncoordinated drainage patterns found over much of the Australian continent. The problem of modelling the natural spatial variability in drainage density, with the medium to coarse resolution spatial data usually available at continental scales, is also unresolved. Readily apparent, however, is the critical importance of DEM quality. Accurate representation of surface drainage characteristics, using drainage informed elevation gridding routines such as ANUDEM, removes the need for extensive DEM pre-processing.

Section 3.2.2 evaluated a number of schemes for sub-dividing and coding drainage basins. The utility of the efficient numbering scheme employed by the Pfafstetter system, and its application internationally, recommends its use for a new Australian stream and catchment reference system. Adoption of this coding scheme dictates the manner in which drainage basins are sub-divided. The applicability of this scheme to the distinctive drainage structures found in Australia will be explored in Chapter 5.

Chapter 4. A new method of drainage analysis and its application

None of the Australian catchment frameworks meet the requirements of a spatial framework for continental scale conservation planning. In Chapter 3 it is recommended that a new nested stream and catchment framework be developed. These are commonly derived by drainage analysis of a DEM. However, the suitability of existing methods for continental scale application is questioned, in particular whether these were applicable to the diverse drainage structures found in Australia. It was also noted that extensive pre-processing could be avoided if greater attention was given to accurate representation of drainage structures in the DEM.

Here, a new method of drainage analysis is developed that overcomes many of the limitations of the existing methods. Its application in deriving a stream network and associated sub-catchments is demonstrated. The accuracy of the drainage analysis products is assessed to determine their applicable scale and suitability for various conservation planning tasks. The development of a multi-scale nested catchment framework that completes the new stream and catchment reference system is described in Chapter 5.

An overview of the new drainage analysis procedures is presented in Section 4.1. and the details of their application and testing in Section 4.2. Section 4.3 reports on the new stream network and an evaluation of its accuracy. These results are discussed in Section 4.4, focussing on a) the methodological contribution of the new drainage analysis procedures, their limitations and opportunities for future enhancement and b) the properties of the derived stream network.

4.1. Introducing a new method of drainage analysis

The new drainage analysis procedures are appropriate for broad scale application with medium to coarse resolution (e.g. 80m to 2.5km) DEMs. They specifically address the key concerns about existing methods that were raised in Chapter 3, namely:

- DEM preparation – ensuring the DEM accurately represents surface drainage characteristics
- Channel initiation – determining the extent of the derived channel network
- Accommodating uncoordinated and distributary drainage structures – enabling flow bifurcations and connecting anabranches
- Delineating catchments that recognise connections via distributaries

Box 4.1 sets out the critical steps in the drainage analysis. An overview of the generic features of the procedures follows. Application specific details are demonstrated via their continent-wide application in Section 4.2.

Box 4.1 Drainage analysis steps for stream and catchment delineation

- 1. Develop a drainage enforced DEM**
- 2. Trace the DEM-derived flow pathways from the head of mapped streamlines to delineate a preliminary stream network**
- 3. Add flow directions to major anabranches and re-trace the flow pathways to delineate the anabranching stream network**
- 4. Remove unchannelised sections from the DEM-derived stream network guided by mapped streamlines**
- 5. Delineate drainage basins and sub-catchment areas for individual stream segments**

4.1.1. Step 1: DEM preparation

The DEM underpins the drainage analysis and in large part determines the accuracy of drainage derivatives. Thus, DEM development is viewed as a critical component of the drainage analysis and the first step in the procedure (Box 4.1). By accurately representing surface drainage features within the DEM rather than attempting to superimpose them later, DEM pre-processing steps such as sink filling and stream burning (Section 3.2.1.3) are avoided. Developing accurate DEMs depends on comprehensive correction of DEM source data errors and inadequacies, and drainage enforcement to ensure connected drainage structure.

4.1.2. Step 2: Channel initiation

Identifying appropriate channel initiation thresholds to delineate stream networks with sensible drainage density characteristics is an essential step in most drainage analysis procedures (Section 3.2.1.1). This task is particularly problematic at continental scales where it is rarely possible to use the high resolution (10m or finer) DEMs that are necessary for adequate description of hydrological and geomorphological properties relevant to channel initiation (Montgomery and Foufoula-Georgiou, 1993; Zhang and Montgomery, 1994; Quinn *et al.*, 1995). In most cases, the real thresholds of channel initiation are well below the resolution of medium to coarse resolution continental DEMs.

However, the streamlines shown on topographic maps (the “blue lines”) accurately identify the extent of channelised flow for a given map scale, though potentially subject to variable cartographic interpretation. The channel heads of these mapped streams provide an obvious

solution to the problem of channel initiation. They are identified automatically from digitised and directed streamlines. Directed streamlines are critical to many drainage analysis tasks that require accurate identification of network topology, including development of accurate drainage enforced DEMs (Section 4.1.1). They require each stream vector to be oriented so that its vertices are arranged in sequence in the direction of flow. Channel heads are then identified simply from the arc-node topology of the stream vectors, that is, as those nodes (endpoints) that occur as a start (“from”) node but never an end (“to”) node. Tracing the DEM-defined flow pathways, from the grid cells in which the channel heads are located to a terminal sink or the river mouth, maps the stream network (Figure 4.1). Flow pathways are defined using a single flow direction algorithm. This, the second step in the drainage analysis procedure (Box 4.1), requires streamline mapping at a scale matching that of the DEM.

4.1.3. Steps 3 and 4: Distributary and uncoordinated drainage structures

Distributary drainage systems that bifurcate and route to more than one terminal point or outlet are not represented in the stream network delineated at step 2. One arm of the bifurcating channels is omitted because the traced flow path is defined with a single flow direction algorithm (Figure 4.2). With very high resolution DEMs this problem may be overcome by judicious application of a multiple flow direction algorithm (Section 3.2.1.2). However, the horizontal and vertical resolution of continental scale DEMs are usually insufficient to reliably direct bifurcating flow according to elevation differences. Again, the digitised version of streamlines, mapped at a scale consistent with that of the DEM, provides the basis for a

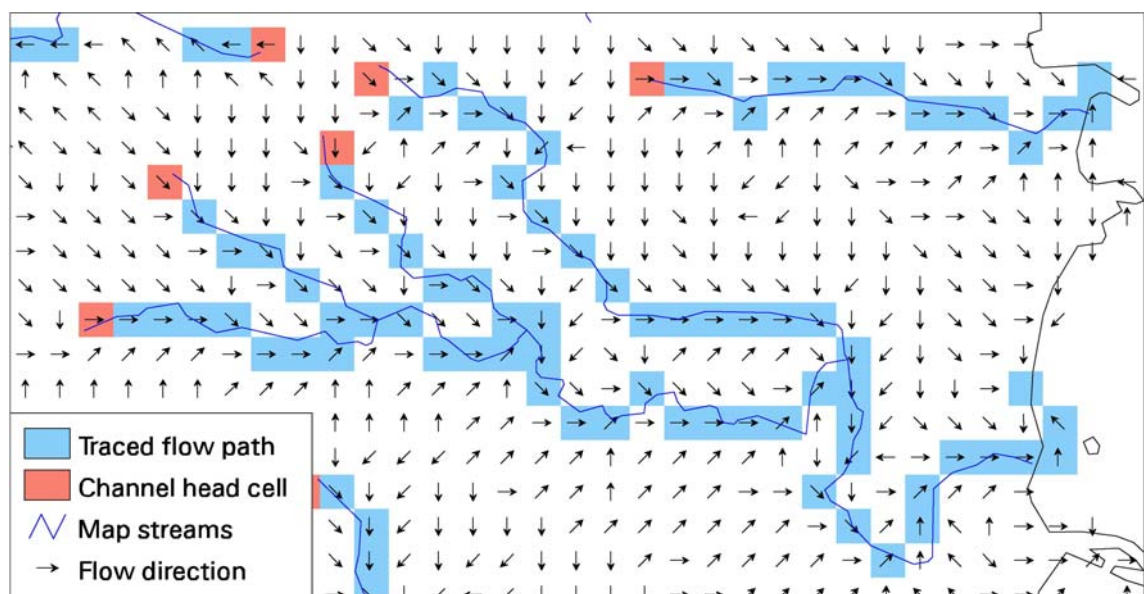


Figure 4.1. Delineating a stream network step 2: identifying channel heads from mapped streamlines and tracing flow pathways. Grid cells with no flow direction shown are terminal sinks.

workable solution.

A bifurcation in the channel network is identified from the topology of a directed stream network. An additional flow direction to the alternative channel is added to a grid cell on the interim stream network from step 2 that is closest to this flow bifurcation (Figure 4.2). The stream network is again traced from the channel heads to its terminal cell(s) but this time the flow path diverges at each of the coded bifurcations. A distributary channel system may bifurcate many times, so this process of adding a flow direction and redefining the stream network is repeated until all desired stream bifurcations are coded with two flow directions and the stream network is fully defined. The stream network is developed iteratively in this way so that bifurcations are not coded for grid cells that are bypassed by the DEM-defined flow path. This could occur where there is a difference in stream placement between the maps and the DEM representation due, for example, to generalization of the input streamline data by the elevation gridding program.

Mirroring the complex channel patterns found in extensively anastomosed and braided systems with coarse resolution, continental-scale DEMs is unrealistic. Channels will branch and rejoin across an area often no more than that of one or two grid cells. So that bifurcations are coded only for major anabranches and distributaries that connect across catchments, two rules are applied. The first rule stipulates that a second flow direction is added only to those grid cells where the contributory areas of the bifurcating channels do not substantially overlap. To implement this rule, sub-catchments are seeded in the standard way by a stream network delineated with a contributing area threshold. An additional flow direction is then added where the bifurcating channels flow into different sub-catchments. The value of the contributing area threshold used determines the number of bifurcations recognised under this rule – the larger the threshold the smaller the number of bifurcations that will be coded. The second rule specifies that bifurcations be coded wherever the individual channels have different names.

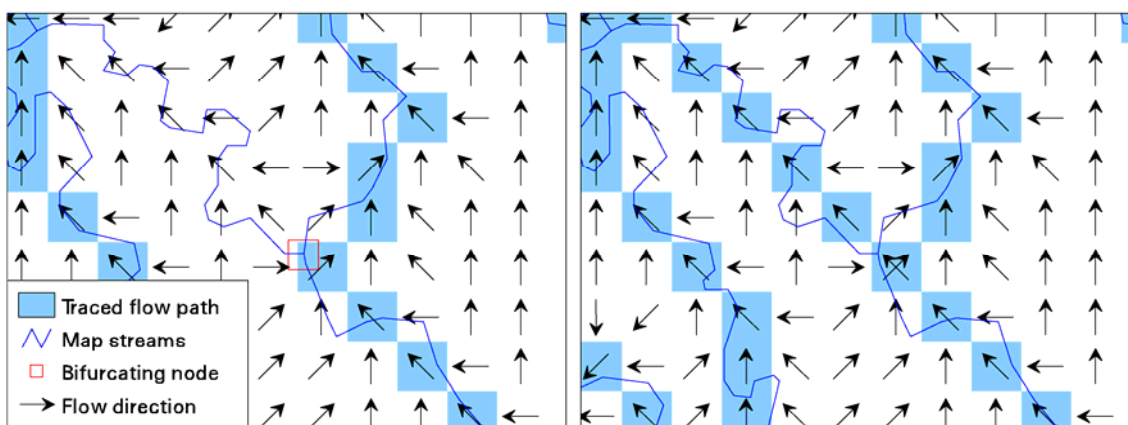


Figure 4.2. Delineating a stream network step 3: adding major anabranches. Missing anabranch in flow path traced with D8 flow directions (left); flow direction added at bifurcation and stream network re-traced to include anabranch (right)



Figure 4.3 Delineating a stream network step 4: removing stream segments greater than a threshold flow path distance to a mapped streamline

This is step 3 of the drainage analysis procedure (Box 4.1) and produces a fully connected stream network including major anabranches. In extremely arid regions however, runoff may be insufficient for the maintenance of a connected channel system across the valley floors. Elsewhere tributary channels may terminate in wetlands or deltaic distributary systems of channels. Hence, the last step in the development of the stream network (Box 4.1) is to generate a more realistic representation of the actual channelised flow network by breaking the derived stream network where there are significant disjunctions in the mapped streams and where a stream enters a large lake. Criteria based on the flow path distance to the next mapped streamline and lake area are used to decide where the stream network should be broken. Thus, if the next mapped streamline is more distant than the flow path threshold or the area of a lake is greater than the specified minimum lake area, the corresponding sections of the derived stream network are removed (Figure 4.3).

4.1.4. Step 5: Delineating catchments

The final step in the drainage analysis procedure (Box 4.1) is to delineate drainage basins and component sub-catchment areas for each element of the stream network. This is usually a straightforward task. The catchment is defined by the grid cells in the flow path upstream of the

selected outlet or catchment seed and labelled with a unique numeric identifier using GIS tools such as the Arc/Info Grid WATERSHED function (ESRI, 2004).

However, it is not clear to which catchment seed the grid cells upstream of a flow bifurcation should be assigned. As most applications assume a grid cell is a member of just one catchment unit, the catchment delineation algorithm of the new drainage analysis procedures must choose between the alternative catchment seeds, in the simplest case, by labelling the upstream grid cells with the catchment seed identifier having the higher value. In the special case of drainage basins, a single identifier is assigned to all connected drainage areas even where a drainage basin has more than one terminal outlet or sink.

4.2. Applying the new method

Figure 4.4 sets out the major steps involved in applying the drainage analysis procedures to produce a new stream and catchment reference system for Australia. Assembling the crucial supporting data layers, including a suitable DEM, associated flow direction grid and matching stream vector coverage, is the first stage in the process. Gridding of the streamline arcs and their nodes is necessary for the grid-based drainage analysis procedures. Two types of node are identified: i) those that represent the channel heads which initiate the stream network trace and ii) those that identify stream bifurcations where additional flow directions will be added so that the anabranching stream network may be traced. At this stage, shortcomings in the available flow direction grid (crossing flow paths, tails and spurious sinks) also need to be rectified. Finally, the anabranching stream network is clipped back to that which could be reasonably supported by the DEM resolution and sections of the network are removed where topographic mapping indicates there is no channelised flow. Details of each step are given below. A parallel set of procedures, enabling drainage basins to be delineated and successively sub-divided according to the Pfafstetter system, is described in Chapter 5.

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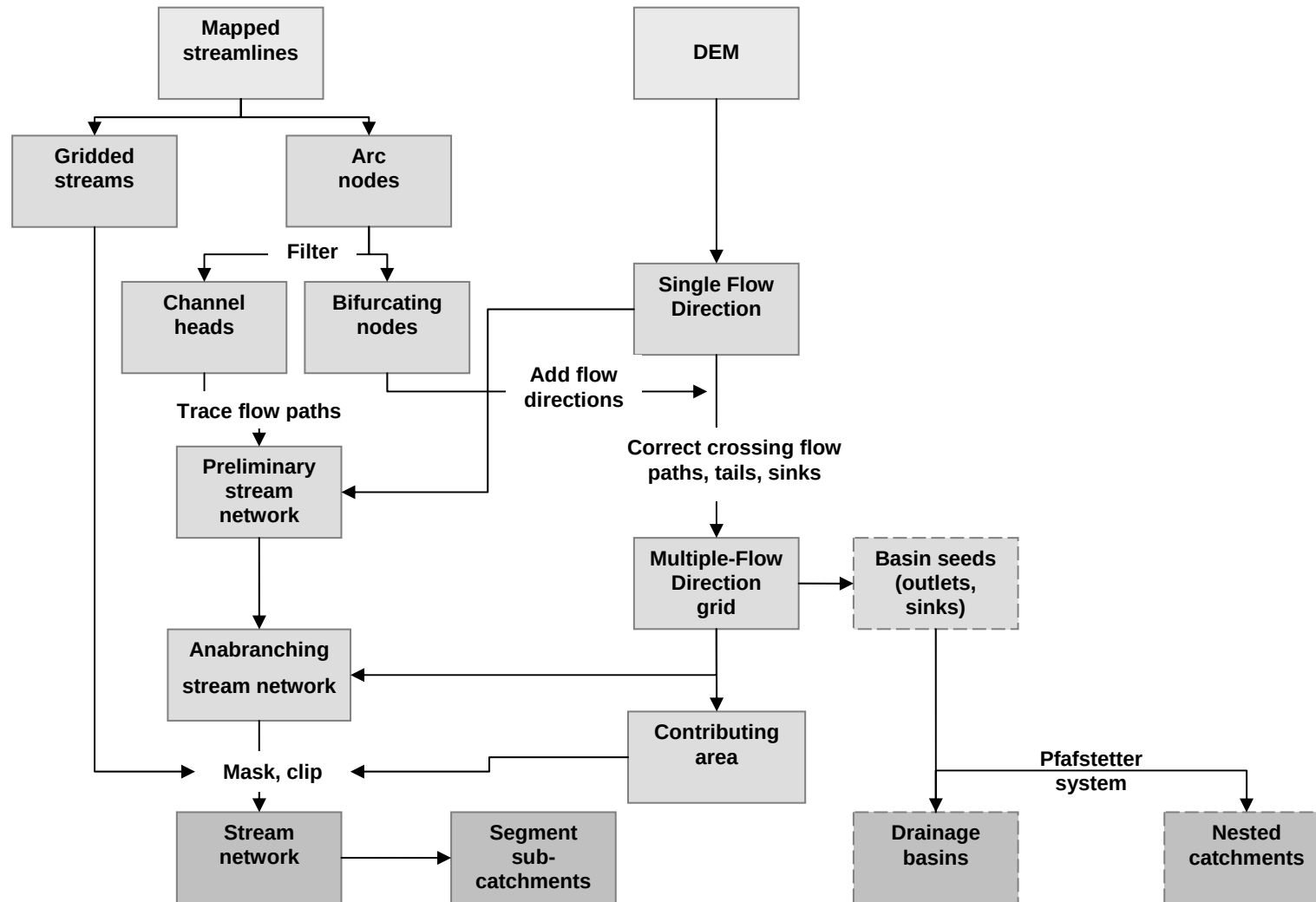


Figure 4.4 Developing a new stream and catchment reference system for Australia. Dashed lines indicate steps covered in Chapter 5

4.2.1. Supporting data layers and critical functions

The new drainage analysis procedures and the routines used to derive supporting data layers were programmed in the Arc/Info Macro Language (AML), a high level algorithmic language (ESRI, 2004). Programs coded in FORTRAN90 were developed for essential flow accumulation, routing and catchment delineation tasks, as the equivalent Arc/Info functions only operate for single (D8) flow direction grids. Some procedures were also written with Unix shell commands. The Arc/Info Workstation GIS software (Versions 7.0 to 8.1) (ESRI, 2004), particularly its Grid and Arcedit modules, were used extensively for editing, analysis and display. Maps were also produced within the ArcMap application of the ArcGIS Desktop software (ESRI, 2005).

4.2.1.1. The DEM

The national 9-second DEM (Geoscience Australia, 2001a; Hutchinson *et al.*, 2001) is the highest resolution, drainage enforced DEM with continental coverage available for Australia. The grid spacing of 9 seconds of latitude and longitude equates to a distance on the ground of between 194m and 265m in an east-west direction (depending on latitude) and about 270m in a north-south direction. Grid cells range between 5.6 and 7.7 ha (mean 7.0 ha) in area. A substantial upgrade of the version 2 DEM by the Centre for Resource and Environmental Studies (CRES) at the Australian National University (Hutchinson *et al.*, in prep.) (Figure 4.5) was completed and made available for this project in March 2004. The upgrade particularly focussed on the improved representation of drainage divides, systematic inclusion of cliff lines and lakes and further correction of source data errors (Table 4.1). Over 22,000 true sink or depression points, identified from 1:100,000 scale topographic maps, were represented in the DEM. They were most often found in sand dune fields, karst areas and ephemeral lakes and clay pans in low relief areas in arid or semi-arid regions.

The upgrade utilised an enhanced version of the ANUDEM elevation gridding program (Version 5.1) (Hutchinson, 2004a). As described earlier (Section 3.2.1.3), the ANUDEM program applies drainage enforcement and incorporates directed streamline source data guaranteeing derived flow pathways are consistent with input drainage networks. Like the previous upgrade (Hutchinson *et al.*, 2001), the DEM was gridded in overlapping tiles corresponding with the geographic extent of Geoscience Australia's 1:1 million scale topographic map sheets, using an iterative procedure that involved extensive checking, editing and re-gridding. Program generated diagnostics, including spurious (non-data point) sinks and large residuals were inspected for potential errors and inadequacies in the DEM source data by comparison with digital 1:100,000 scale topographic mapping. Sinks, in particular, are a very

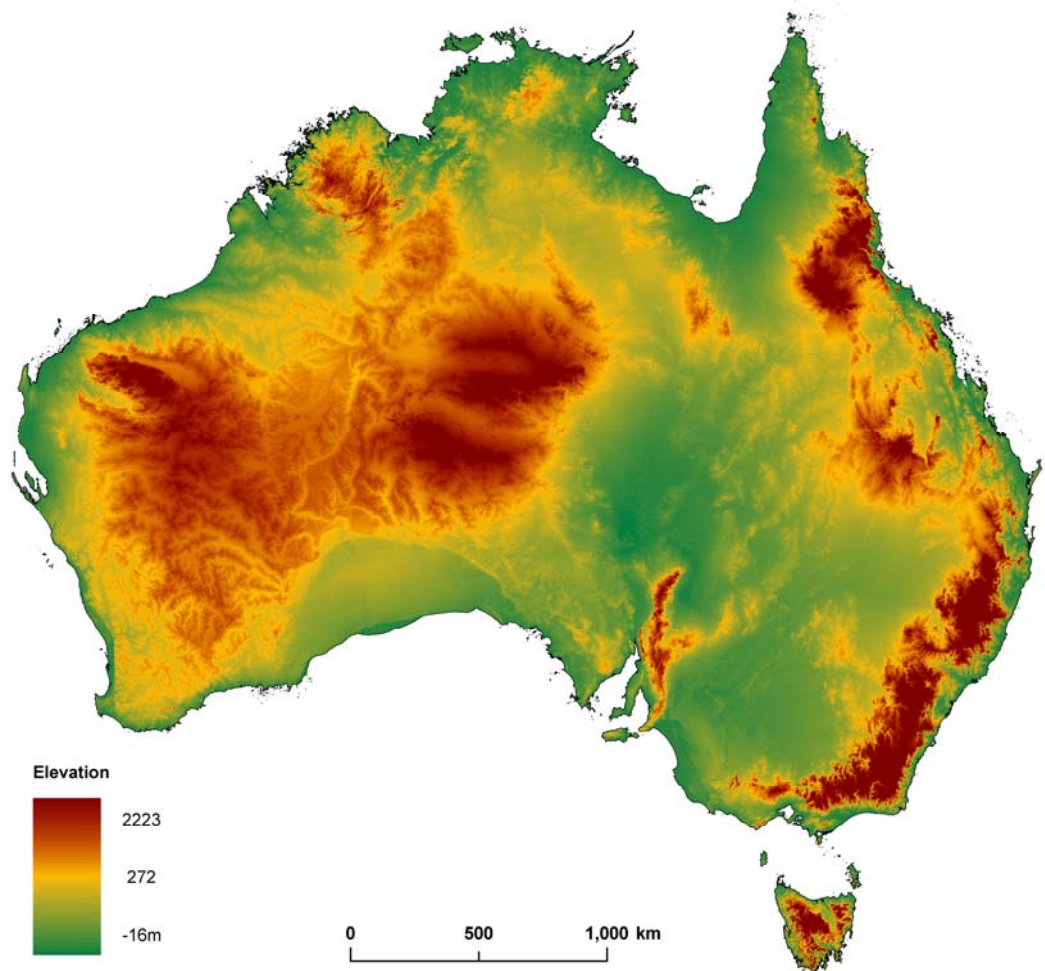


Figure 4.5. The 9 second DEM version 2.1. Elevation values are mapped with a continuous colour ramp and a linear stretch of values between the mean plus or minus two standard deviations.

sensitive and practical indicator of deficiencies in land surface representation (Hutchinson and Gallant, 2000). Basin boundaries derived from the DEM were also compared with those of the established AWRC drainage basins (AUSLIG, 1997) to identify locations where extra source data were needed to reliably define drainage divides. The individual DEM tiles were joined together with the Arc/Info Grid MOSAIC function. This function uses a proximity analysis algorithm to create a smooth transition in cell values across the overlap zone and minimise abrupt boundary changes between tiles (ESRI, 2004).

Hutchinson *et al.* (2001) judged the scale of the 9-second DEM Version 2 to be about 1:250,000, making it suitable for applications at regional to continental scales. As the density of source data is further improved (Table 4.1) the DEM Version 2.1 is confidently considered to be applicable at scales of 1:250,000 or finer and thus appropriate for the task of delineating a new national stream and catchment reference system for Australia.

Table 4.1. Source data for the 9-second DEM with numbers of corrections and additions for version 2.1. Data added to overcome deficiencies in the original source data were digitised from 1:100,000 scale topographic maps.

Data type ₁	Additions	Corrections	Total	Data source
Sink points	2,024		22,166	1:100,000 scale topographic maps
Directed streamline arcs	48,573 ₂	2,018	1,940,066	TOPO250K GEODATA Database Version 1 (AUSLIG, 1992b)
Elevation spot heights	151,565	17,052 ₃	4,929,577	TOPO250K GEODATA Database Version 1 (AUSLIG, 1992b), National Geodetic Data Base (NGDB) and Radar altimeter data for Lake Eyre (supplied by AUSLIG)
Lakes and reservoir polygons	119,474		119,474	TOPO250K GEODATA Version 1 (AUSLIG, 1992b)
Cliff line arcs	23,235	28	23,839	TOPO250K GEODATA Version 2 (Geoscience Australia, 2003c), 1:100,000 scale topographic maps
Coastline	All arcs and polygons defining the Australian coastline			GEODATA COAST-100K (AUSLIG, 1992a), coastal inlets from the GEODATA TOPO-250K coastline (AUSLIG, 1992b)

1. See the 9 second DEM user guide for details (AUSLIG, 2001)

2. Includes 10,122 drainage lines digitised to assist drainage enforcement

3. Includes 11,244 spot heights on sand ridges that were removed for more consistent gridding

4.2.1.2. Flow direction

A grid of flow directions was derived from the grid of aspect values produced by the ANUDEM program for each of the overlapping DEM tiles. The aspect values incorporate the information provided by the directed streamline input data and program imposed side conditions. Stein *et al.* (2002) found this aspect grid provided a better representation of surface flow paths in areas of low relief than would be achieved by simply allocating the direction of steepest descent as in the D8 method.

Aspect values (in degrees from north) were converted to values of the easterly and northerly components of the unit vector, computed by taking respectively, the sine and cosine of the direction (transformed to radians). Component values could then be merged using the Arc/Info Grid MOSAIC function, avoiding abrupt changes at tile borders. Finally, the angle of flow direction was computed as the inverse tangent of the ratio of the merged northerly to easterly

32	64	128
16		1
8	4	2

Figure 4.6. Flow direction codes. The centre cell is assigned the value shown for the cell in the direction of flow. For example, if the direction of flow of the centre cell was to the east the cell would be coded with a value of 1.

components and reclassified into the standard D8 codes used by Arc/Info (Figure 4.6) using the RECLASS function in Arc/Info Grid and a lookup table.

In areas of very low relief where surface flow paths were not well defined by the DEM, this process occasionally produced closed loops in the overlap zones. Loops complicate subsequent analysis. Some Arc/Info functions, for example, do not complete if the flow direction grid contains a loop. Thus the flow direction code of one cell in each loop was converted to a sink code (a value of 0).

A relatively small number of spurious (i.e. non-data point) sinks remained in the DEM. These might have been removed by another iteration of the DEM development procedures but this would have taken considerable time. Instead, they were cleared with an automatically applied procedure that altered the flow direction codes directly. The sink cell was preferentially directed to a neighbouring stream cell (i.e. one through which a map stream passed) or otherwise to the lowest neighbouring cell that was not within the catchment of the sink. Neighbouring cells are adjacent cells in any of the eight directions. To clear the few spurious sinks not cleared in this way, the flow direction codes of cells in the vicinity of the sinks were manually altered with the Arc/Info GRIDEDIT functions using 1:100,000 scale topographic maps to guide corrections. Elevation values were not altered. Automated sink removal procedures (Section 3.2.1.3) were not adopted because the flow direction grid would then need to be recomputed directly from the DEM thereby losing the more accurately encoded surface flow paths produced by ANUDEM.

Other shortcomings in the flow direction grid produced crossing flow paths or catchment “tails”. These were corrected with modified versions of procedures developed for an earlier drainage analysis project (Hutchinson, Stein *et al.*, 2000). Crossing flow paths arise when the inter-cardinal flow directions of adjacent grid cells differ by 90 degrees (e.g. 45° and 315° or 135° and 225°) (Figure 4.7). They cause the Arc/Info Grid STREAMLINE function, used to derive a vector representation of a gridded stream network, to terminate with errors and

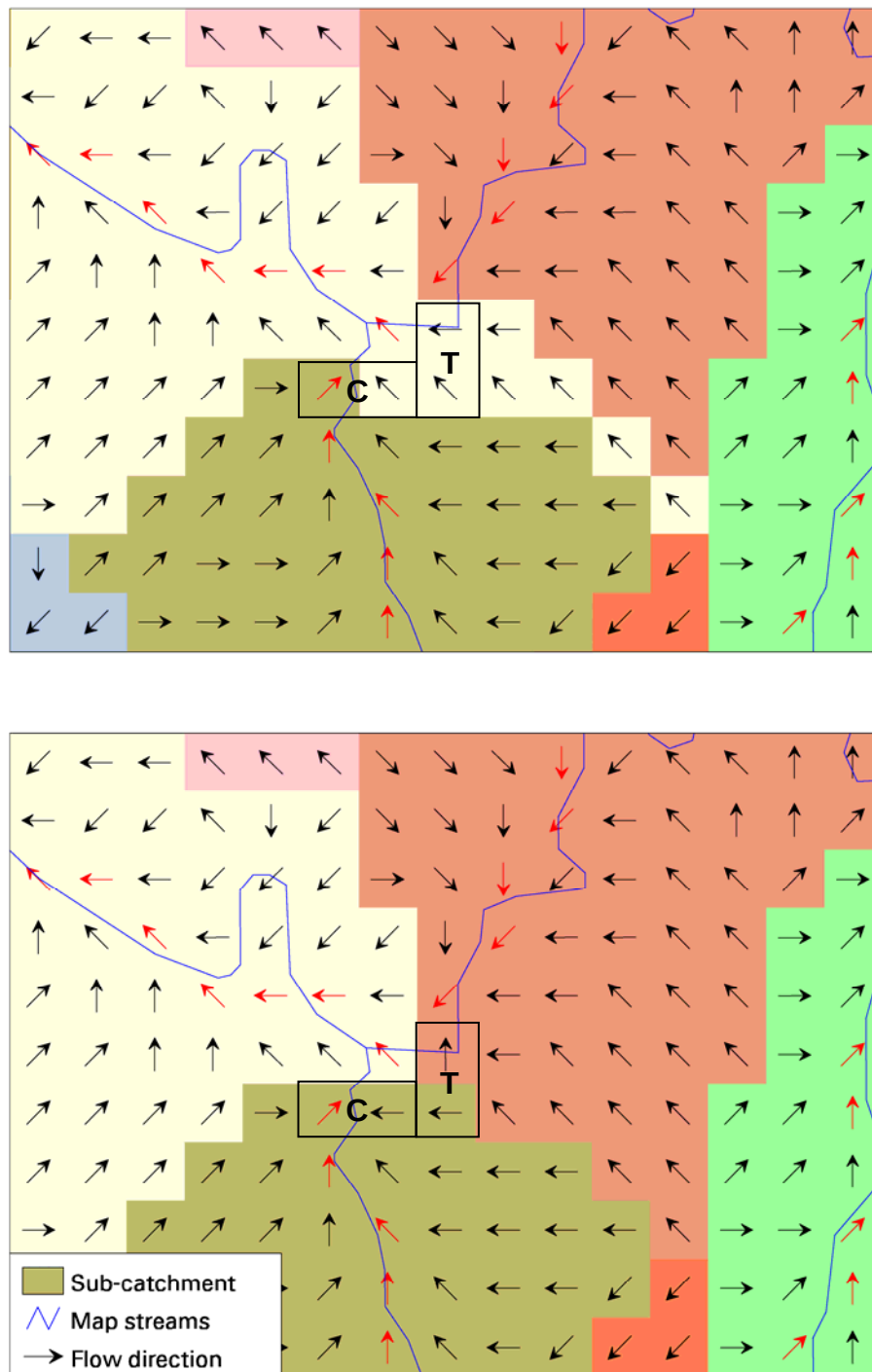


Figure 4.7 Correcting flow directions to remove crossing flow paths and sub-catchment tails: before (top) and after (bottom) correction of flow direction codes. The flow directions in red show the derived stream network. Adjacent cells at C, producing crossing flow paths, were corrected by changing the flow direction of the cell adjacent to the stream. To remove the catchment “tails” the flow directions of the two cells at T are altered so that they and the cells upstream are directed towards the outlet of the adjacent tributary stream segments.

may produce “tails” when sub-catchments are delineated. Tails are zones of a sub-catchment that lie above an upstream confluence so reducing sub-catchment spatial contiguity. Crossing flow paths on or immediately adjacent to the derived stream network were corrected by re-directing the flow direction of one of the crossing cells to the lowest neighbouring grid cell (Figure 4.7). The flow direction of grid cells not included in the traced stream network was altered preferentially. Where the traced network included both of the crossing cells, the flow

direction of the cell with the higher elevation was redirected. Other arrangements of flow directions will also produce catchment tails. Wherever possible, the connectivity of sub-catchments was improved by altering the flow direction codes of grid cells in the tail so that they were routed to the outlet cell (seeds) of the upstream tributary sub-catchment in the direction of greatest slope (Figure 4.7), providing that this outlet cell had a lower elevation than the cells upstream of the altered cell and that closed loops or sinks were not introduced by the changes. Corrections were applied after step 3 in the drainage analysis procedure (Box 4.1) to ensure they took into account the added flow directions at flow bifurcations.

4.2.1.3. Mapped streamlines

Mapped streamline vectors were extracted from the GEODATA TOPO 250K version 1 database (AUSLIG, 1992b). This database captures all streamlines shown on the 1:250,000 scale topographic map series with a length of at least 2.5 kilometres. It also includes a number of floodplain features such as billabongs or flood runners and distributaries. Extensive editing and correction of these streamlines was undertaken for DEM development. Streamlines arcs that erroneously crossed over a ridge were split and canals incorrectly attributed as streamlines removed. Each stream arc was oriented so that the vertices were arranged in a sequence in the direction of flow, as required by ANUDEM. Billabongs and other floodplain features were deleted where they were found to create difficulties for the DEM interpolation. The GEODATA streams were supplemented as necessary for more accurate definition of drainage divides with streamlines digitised by CRES from 1:100,000 scale topographic maps (Table 4.1) (Hutchinson *et al.*, 2001, in prep.).

Streamline vector coverages were appended and gridded at 9-second resolution with the Arc/Info Grid LINEGRID function. Grid cells were assigned the value of the unique arc identifier. Where more than one arc falls within a grid cell, the identifier of the longest arc within the cell was allocated. This identifier links the gridded stream arcs to the original arc attribute tables, providing access to additional attributes such as stream name and arc node numbers.

4.2.1.4. Accumulating flow or other attributes and apportioning flow

Many applications require flow or other attributes to be accumulated or routed downstream according to the flow paths specified in the flow direction grid. This is normally a simple task that can be accomplished with standard GIS functions. For example, the Arc/Info Grid FLOWACCUMULATION function will do this but it assumes the D8 method of flow direction and treats the multiple flow directions at flow bifurcations as sink cells. Consequently, FORTRAN 90 programs were written to perform this function. These programs implement the

very efficient recursive algorithm of Mark (1988), extended to accommodate multiple flow directions using the FORTRAN 90 bit operators. At flow bifurcations the accumulated value is passed to neighbouring cells in both of the directions coded by the flow direction value, apportioned either so:

- both downstream cells receive the full value of the upstream cell or
- each downstream cell receives a specified proportion of the value of the upstream cell

The first option was designed for the calculation of catchment area, defined to be all of the upstream land and water surface area that drains to a particular location (Gordon *et al.*, 1992). Its coding required special treatment to ensure catchment areas are not counted twice if bifurcating streams rejoin further downstream. The second option is used for most other flow related attributes where it is appropriate to divide the accumulated “flow” downstream of the bifurcation. It requires an additional input grid specifying the appropriate proportions.

Proportions were allocated according to stream naming conventions based on the observed values of median annual runoff for a national set of gauging stations compiled for the NLWRA (Peel *et al.*, 2000) and professional judgement. The median annual runoff of 193 gauging stations on rivers (216 mm) was found to be exactly twice that for the 125 gauging stations on creeks (108 mm), suggesting “flow” should be apportioned between “rivers” and “creeks” in the ratio of 2:1. The value for the proportion of flow that should be diverted to unnamed streams at a flow bifurcation was based on personal observations, as none of the gauging stations were located on unnamed streams. Diverting to unnamed streams one quarter of that apportioned to “creeks” or one eighth that to “rivers” produced results that reasonably reflect relative stream sizes. Accordingly, upstream values were apportioned in the ratio of 8 rivers: 4 creeks: 1 unnamed stream. Where information is available for a particular distributary system, observed values of flow distribution could be readily substituted for the simple rules used here.

4.2.1.5. Pour-points

Attribution of stream segments is done by considering the attribute value at the segment pour-point or outlet cell, that is, the cell on a stream segment immediately above a confluence that directly flows into the cell of a downstream segment (Figure 4.8). For terminal segments, the pour-point is the most downstream cell in the segment. This is often a sink or otherwise an outflow to the ocean. Pour-point cells were identified and stored as a separate grid layer.

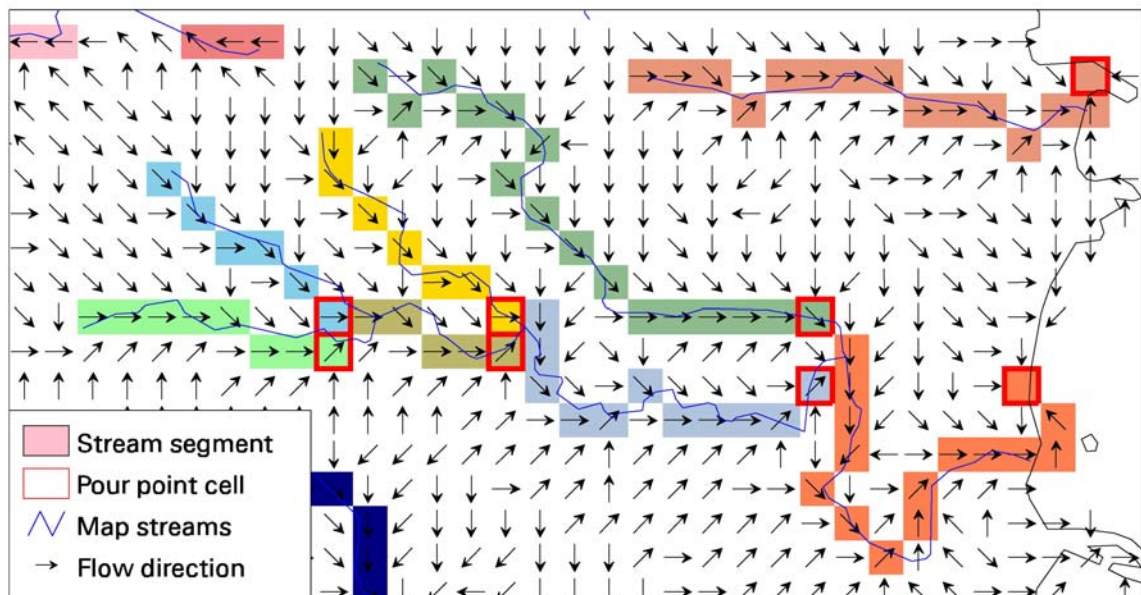


Figure 4.8. Segment pour-points. Stream segments are shown as grid cells shaded with the same colour.

4.2.1.6. Stream order

Stream order is a commonly adopted descriptor of catchment size and an indicator of network position (e.g. Smith and Kraft, 2005). The system attributed to Strahler numbers all source channels as order one and only increments order downstream of a confluence between two streams of the same order (Gordon *et al.*, 2004). It is relatively insensitive to changes associated with tributary inputs, but nonetheless is widely used and understood and remains a useful basis for description of stream network characteristics (Knighton, 1998).

The stream order method is intended for connected drainage networks and is not sensibly applied to uncoordinated drainage patterns. Anabranching stream networks also require special consideration. Stream order was derived for the connected network produced at step 3 of the drainage analysis procedure and subsequently attributed to segments in the final drainage network. It was computed with a FORTRAN90 program that assigned both channels downstream of a flow bifurcation the stream order of the upstream channel and tested for bifurcating channels rejoining before incrementing stream order. The test operated on the source of the channels at a confluence – stream order was only incremented where two tributaries of the same order joined and where the source of each tributary was different. The source is defined as the highest point upstream and derived for each channel by assigning a sequential identifier, in ascending elevation order, to all source cells within a drainage basin and tracing the maximum identifier downstream.

4.2.2. Delineating a stream network

4.2.2.1. Channel initiation

The locations of channel heads for stream network initiation were inferred from the arc-node topology of the mapped streamlines. Channel heads are the start point of the first order streams in the channel network. They were defined by the set of “from” nodes (i.e. the start point of an arc) that never occur as “to” nodes (i.e. the end point of an arc) (Figure 4.1).

The mapped streamlines also include floodplain features such as billabongs or relict channels that appear to be source channels because they are not connected to other arcs in the network. Most of these were automatically identified and omitted by application of additional criteria that considered the length, name and landscape position of the arcs. Accordingly, the from nodes of unnamed, short (< 5 cells) arcs occurring entirely on a large flat area in a lower landscape position were not considered to be channel heads. Landscape position was determined from the values of the multi-resolution Valley Bottom Flatness Index (mrVBF) and Ridge Top Flatness Index (mrRTF) (Gallant and Dowling, 2003) (Appendix 1), with a threshold value of 3.5 identifying floodplains. Criteria were conservatively defined to ensure that genuine channel heads were not removed inadvertently but therefore did not detect all spurious channel heads associated with floodplain features. Thus channel heads close to major streams were manually checked by reference to 1:100,000 scale digital topographic maps, resulting in the removal of about 5000 more nodes. Major streams were defined as those included in the 1:1,000,000 scale Global Map 1M digital database (Geoscience Australia, 2001b).

After filtering, channel head nodes were gridded at 9-second spatial resolution in preparation for stream network delineation. Occasionally, the from node of more than one stream arc fell into the same grid cell. As the grid cell could be used as a channel head for only one of these streams, the highest elevation stream cell was used as the channel head for the other stream. Using the highest cell as an alternate channel head was also necessary where the flow direction code of the from node cell was routed to another stream arc (Figure 4.9). The preliminary stream network (step 2, BOX 4.1) was delineated by tracing the shortest flow path from the channel heads to a terminal sink or the coast with the Arc/Info Grid COSTPATH function.

4.2.2.2. Coding bifurcations

Channel bifurcations were identified from the arc-node topology of the mapped streamlines. Arc nodes that occur more than once as a from (i.e. start) node indicate a branching channel. The bifurcation was encoded into the flow direction grid by adding the appropriate flow direction code to the grid cell at or adjacent to the cell in which a branching node occurs.

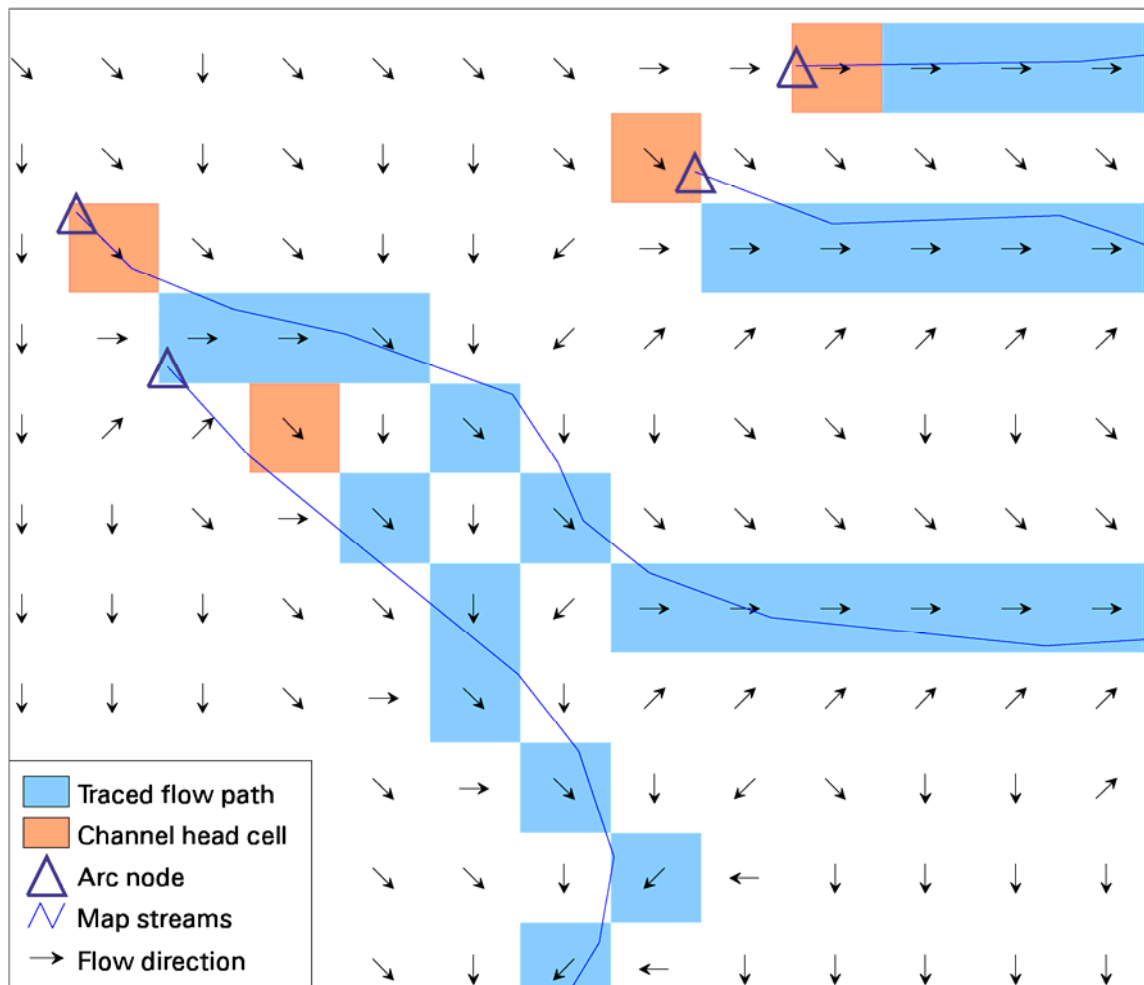


Figure 4.9. Alternative specification of channel heads where the gridded from node of the map streamline arc is captured by the flow path of another stream.

Bifurcating flow directions were only added for major anabranches and distributaries where the name and large catchment membership of the receiving cells satisfied the rules defined previously (Section 4.1.3). Thus, an additional flow direction was added to a grid cell if:

- the grid cell overlayed a branching node or was adjacent to a cell overlayed by a branching node and
- the branching channels did not rejoin directly downstream, that is, they did not share the same to node and
- the added flow direction routed flow to a stream cell with a different name to the downstream cell in the original single flow direction or
- the added flow direction routed flow to a different sub-catchment to the downstream cell in the original flow direction (Figure 4.10)

Stream names were compared using a grid derived by assigning a numerical identifier to each unique name from the name attribute of the mapped streamlines. Sub-catchments were seeded with a stream network delineated using a contributing area threshold of 500km^2 and labelled with the Arc/Info Grid STREAMLINK function. Drainage basins delineated with the Arc/Info Grid BASIN function filled the gaps for drainage areas of less than 500 km^2 . For the most part,

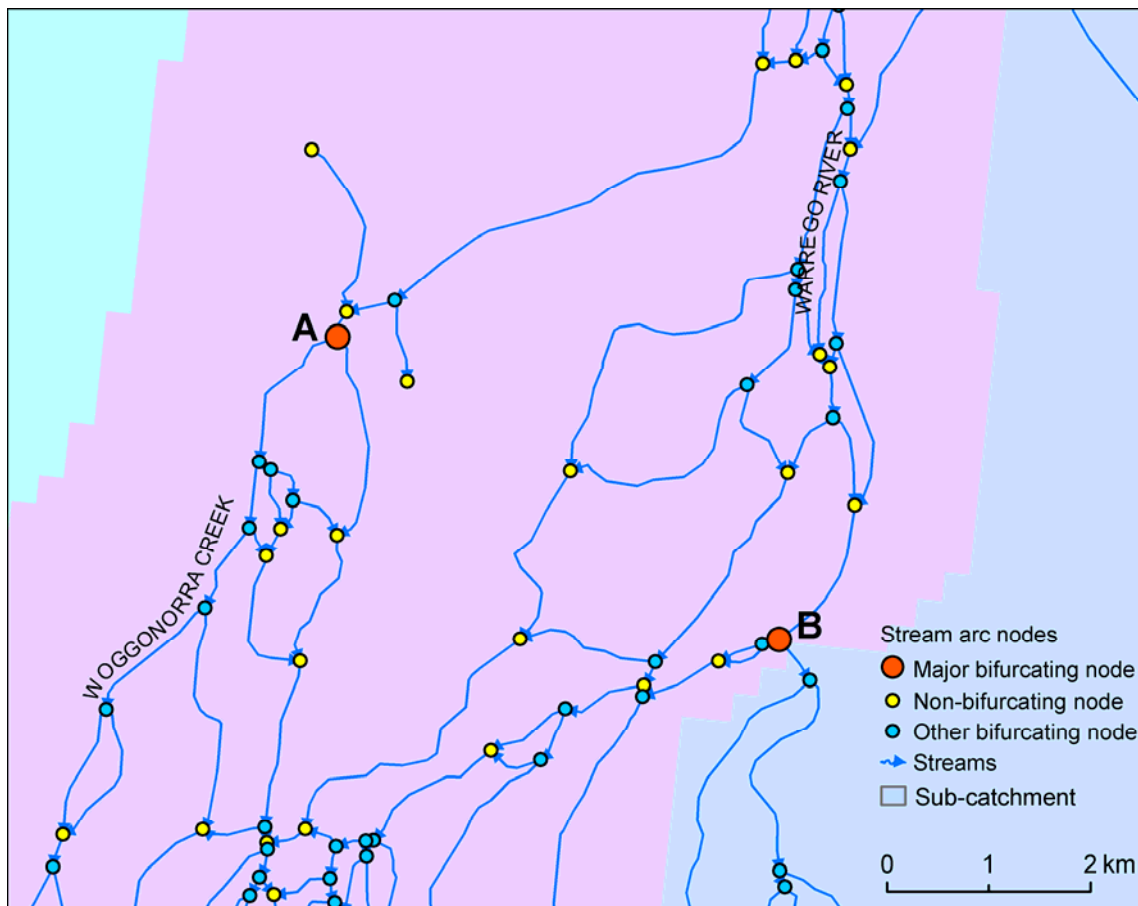


Figure 4.10 Adding flow directions at major flow bifurcations. Flow directions were added at branching nodes where flow is routed to a channel of a different name (A) (Warrego River to Woggonorra Creek) or to a channel draining a different sub-catchment (B). Arrows indicate the direction of flow.

the threshold value of 500km^2 appeared to achieve the desired outcome of selecting the more significant bifurcations.

Program checks ensured that the added flow directions did not create closed loops or sinks in the flow paths and that they routed flow only to streamline arcs that are immediately downstream of the bifurcating node. Downstream arcs were identified from the topology of the mapped streamlines as being arcs with the bifurcating node as their from node. If the immediately downstream arc was too short to be separately gridded the next arc downstream was used. Where flow to more than one cell satisfied all criteria the added direction was that directed to the cell with lowest elevation. Only the highest bifurcation was recognised for each combination of different stream names or sub-catchment numbers. The multiple flow directions were stored as a single digit being the sum of the flow direction codes (Figure 4.6) for the alternate directions. The binary representation of these codes facilitates separation of the component flow directions with bit operators. After coding bifurcations, the new stream network was delineated using a FORTRAN90 program (Section 4.2.1.4) to trace the stream network from the channel heads and the Arc/Info Grid STREAMLINK function to uniquely label the stream segments (step 3, Box 4.1).

4.2.2.3. Finalising the stream network

The resolution of the DEM limits the extent of the stream network and the size of the drainage features that can be extracted. Garbrecht and Martz (1994) established that key drainage features could be reproduced with an accuracy of about 10% when the grid cell area was less than 5% of the network reference area, that is, the average sub-catchment area draining directly into the channel segments. It follows that the smallest catchment area resolvable from a DEM of a given cell size would be 20 times the area of the grid cell area if the derived boundaries were to be accurate to within 10%. Hence, the smallest catchment area resolvable from the 9-second DEM, with a cell area of approximately 0.0625 km^2 , would be 1.25 km^2 . This value also agrees with the figure of about 1 km^2 determined by other studies to be the minimum catchment area reproducible from the 9-second DEM (John Gallant, CSIRO Land and Water, pers. comm. 5/9/02), but is substantially larger than the smallest drainage areas of source channels in the DEM-derived stream network initiated by the mapped channel heads (1 grid cell or 0.0625 km^2). Thus, source channels with a contributing area of less than 1.25 km^2 at their pour-point were removed. Main stem segments, defined as the segments draining the larger upstream contributing area, were retained to their source (Figure 4.11).

The delineated stream network connects all channel heads with a terminal sink or the coast. To more accurately reflect uncoordinated drainage patterns, sections of the stream corresponding to gaps in the mapped streamline network of more than 5km (along the DEM-derived flow path) and overlaying major lakes were deleted and the revised stream segments re-labelled with the Arc/Info Grid STREAMLINK function (Figure 4.3). The 5km threshold value balances the

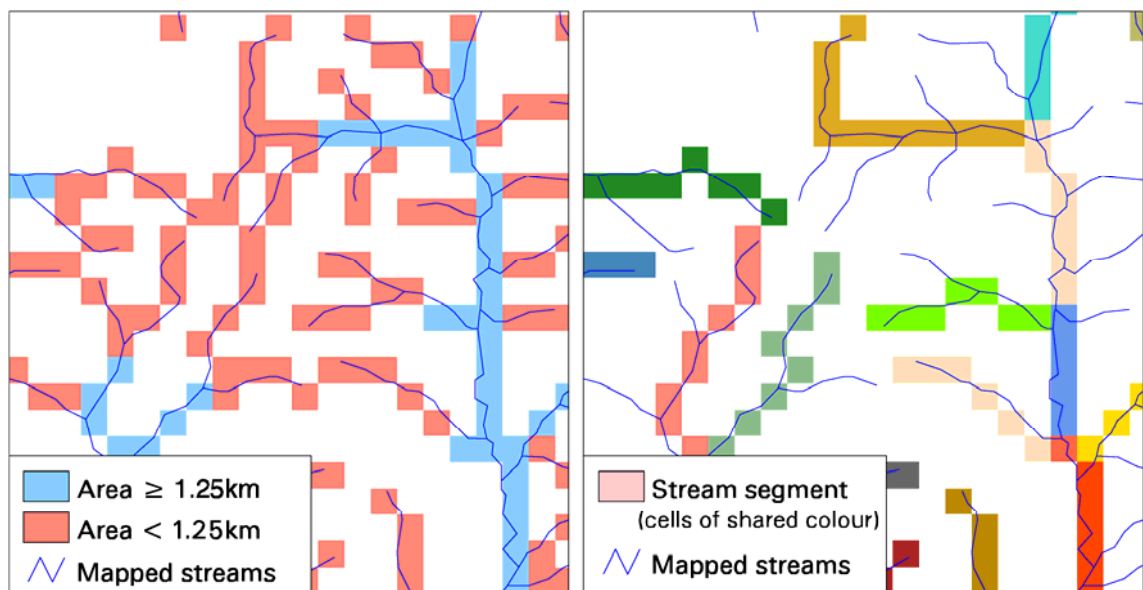


Figure 4.11. Masking the stream network to remove source channels with a contributing area smaller than is reproducible from the 9 second DEM: the preliminary stream network, delineated by tracing flow pathways from channel heads (left) and the final stream network with stream segments indicated by shared colour (right). Compare this network with the truncated network (shown in blue on the left) that would be derived using just a contributing area threshold to delineate the network.

competing requirements of filling small gaps in the mapped streamline network due, for example, to variable cartographic interpretation between map sheets or the presence of local swamps, and accurately representing largely uncoordinated drainage patterns. Major lakes were defined to be natural waterbodies larger than 1.25km² in area that are traversed by a mapped streamline arc attributed as a “connector”. Connectors are artificial drainage lines in the GEODATA database that are used to join sections of the mapped streamline network broken by lakes or reservoirs (AUSLIG, 1992b). Lake polygons were extracted from the GEODATA TOPO-250K database version 2 (Geoscience Australia, 2003c).

The new gridded stream network was visually inspected for obvious errors and systematically checked to ensure it represented all larger streams (streams included in the GEODATA 1 : 2.5 million scale digital database, Geoscience Australia, 2003b). Channel heads, mistakenly omitted by automatic application of criteria to distinguish mapped source channel arcs from floodplain channels, were manually re-inserted. Additional flow directions were added manually at significant bifurcations where the configuration of flow direction codes and stream network placement had prevented automatic coding of the multiple flow direction. After correcting these errors the stream network was revised, though some localised errors were discovered much later in the project and have been noted for future revisions.

Sub-catchment areas were delineated for each stream segment in the derived stream network with the Arc/Info Grid WATERSHED function seeded by the stream network. Grid cells were assigned the unique identifier of the stream segment to which they directly drained. In this case, it did not matter that the WATERSHED function treated the multiple flow directions at flow bifurcations as undefined because these seed cells were already assigned catchment membership. For mapping purposes, a vector representation of the final stream network was derived with the STREAMLINE function in the Arc/Info Grid module.

4.2.3. Validating the drainage analysis

Qualitative assessments of drainage accuracy were undertaken by visual comparison of the derived stream network with the 1:250,000 scale mapped streamlines and the streamlines shown on digital 1:100,000 scale topographic mapsheets at random locations across the country. More objective evidence of drainage accuracy was derived by quantitatively comparing catchment areas and the placement of streams and drainage divides against independent sources.

A comparison of derived catchment areas with independent reference values is commonly used to assess the accuracy of surface drainage pathways and catchment boundaries (Bertolo, 2000; Döll and Lehner, 2002; Vogt, Colombo, Paracchini *et al.*, 2003). Reference values may be taken from published basin areas, but comparisons may be confounded by different definitions of the

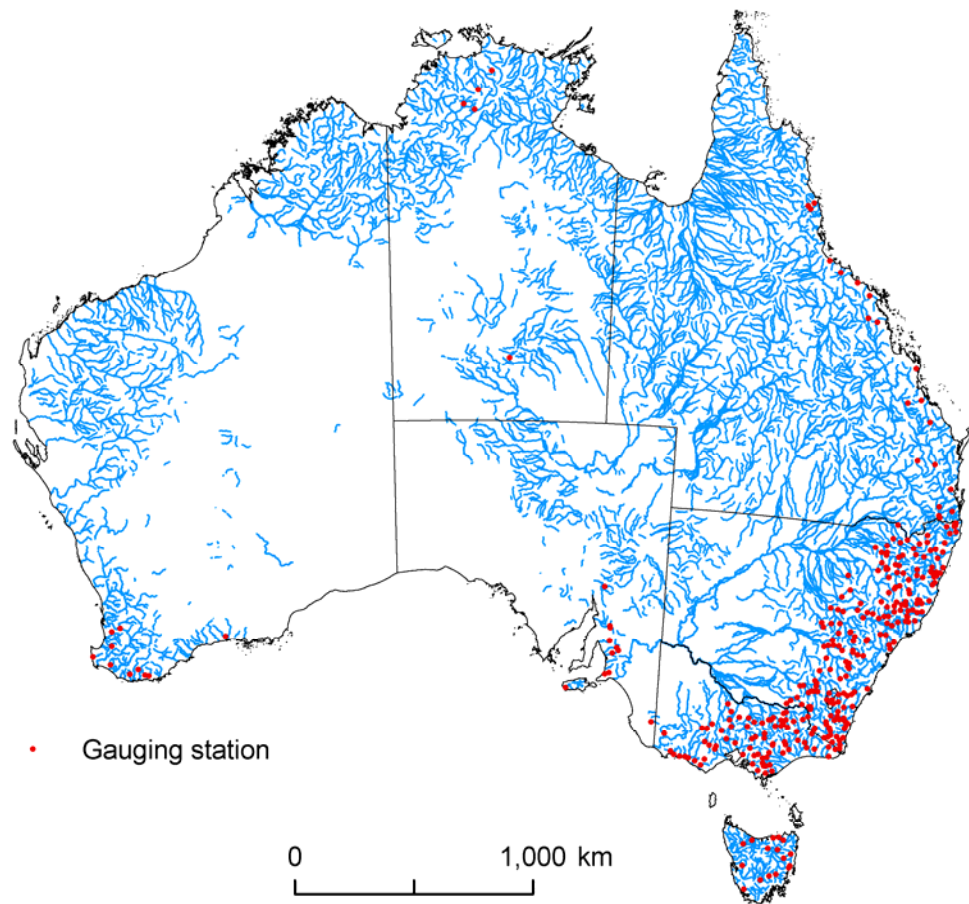


Figure 4.12. The NLWRA streamflow gauging stations used as a reference set to test the accuracy of the DEM-derived catchment areas

catchment outlet or the treatment of arid areas that produce little or no effective runoff (Vörösmarty *et al.*, 2000). Furthermore, the reported areas for Australian basins (e.g. National Land and Water Resources Audit, 2001b) are often based on the AWRC catchment delineations (AUSLIG, 1997) that, as noted earlier (Section 3.1), are not always topographically defined.

A more reliable comparison was possible with the data compiled for the NLWRA Extension of Unimpaired Monthly Streamflow Data project (Peel *et al.*, 2000). This data set, downloaded from the NLWRA Australian Natural Resources Data Library (ANRDL) (http://data.brs.gov.au/asdd/php/basic_search.php), provides the catchment boundary polygons and site locations of 331 streamflow gauging stations across Australia (Figure 4.12). Catchment boundaries had been provided to the NLWRA for gauging stations in Victoria, Tasmania and South Australia and were manually digitised from the 1:100,000 topographic map series for other states (Peel *et al.*, 2000). The reference area for each catchment polygon was derived after transforming the coverage to Lambert Conformal Conic projection co-ordinates. This projection least compromises the representation of both area and shape. Where a second gauging station was located on the same river the area of the catchment polygon for the upstream station was added to that of the lower station. Catchment area for the 15 stations not supplied with a catchment boundary polygon was entered from information provided in Peel *et al.* (2000).

The reference areas were compared to the 9-second DEM-derived catchment area values for the grid cell overlaying the gauging station locations. Due to differences in the scale of base mapping and generalisation of the stream representation in the DEM-derived flow paths, some station locations did not fall on the DEM-derived stream network. These were automatically moved to the adjacent grid cell with upstream catchment area closest to the reference values. Systematic checking to ensure the name of the mapped streamline matched the gauging station description exposed major errors in the co-ordinates provided for the gauging station. For example, stations fell on hillslopes, in the wrong catchment or even in the sea. These errors were corrected, guided by the associated catchment boundary polygons and site descriptions, and the DEM-derived catchment area extracted for all gauging stations.

Small differences in total catchment area can hide large disparities in catchment boundary locations. To test for such discrepancies the gauging station catchment polygons were compared with those automatically delineated from the DEM-derived flow paths using the gauging stations as catchment seeds. Following Vogt, Colombo, Paracchini *et al.* (2003), the proportion of the DEM-derived catchment boundary length that fell within increasingly larger buffers (250, 500 and 1000m) around the reference catchment boundary was computed after transforming catchment boundary layers to Lambert Conformal Conic projection coordinates. Buffers were delineated with the Arc/Info BUFFER command.

To investigate the suitability of the DEM-derived products for finer scale applications, the 9-second DEM-derived catchment boundary for the Cotter River catchment in the ACT was compared to catchment boundaries determined from two high resolution DEMs, a 20m DEM-derived from 1:25,000 scale contour and directed streamline data and a 5m DEM produced from Airborne Laser Scanning (ALS) data (Hutchinson, Stein *et al.*, 2004) (Figure 4.13). The ALS data have a quoted standard elevation error of about 0.1m (Hutchinson, Stein *et al.*, 2004). As before, the proportion of the length of the 9-second DEM-derived catchment boundary falling within specified buffers of the reference boundary location was computed. Boundary co-ordinates were transformed to the Stromlo coordinate system, a local ACT projection based on the Transverse Mercator projection, consistent with the specifications of the high resolution DEMs.

Similar methods were employed to assess the accuracy of the 9-second DEM-derived stream network. Reference data for comparison included the Australia-wide 1:250,000 scale mapped streamlines and for the Cotter River catchment; digitised streamlines from the NSW 1:25,000 scale topographic map series provided by NSW Land and Property Information from their Digital Topographic Database; digitised streamlines from the ACT 1:10,000 scale topographic map series supplied by the ACT Land Information Centre; and streamlines derived from the 5m ALS DEM using a contributory area threshold of 2.5ha (Hutchinson, Stein *et al.*, 2004)

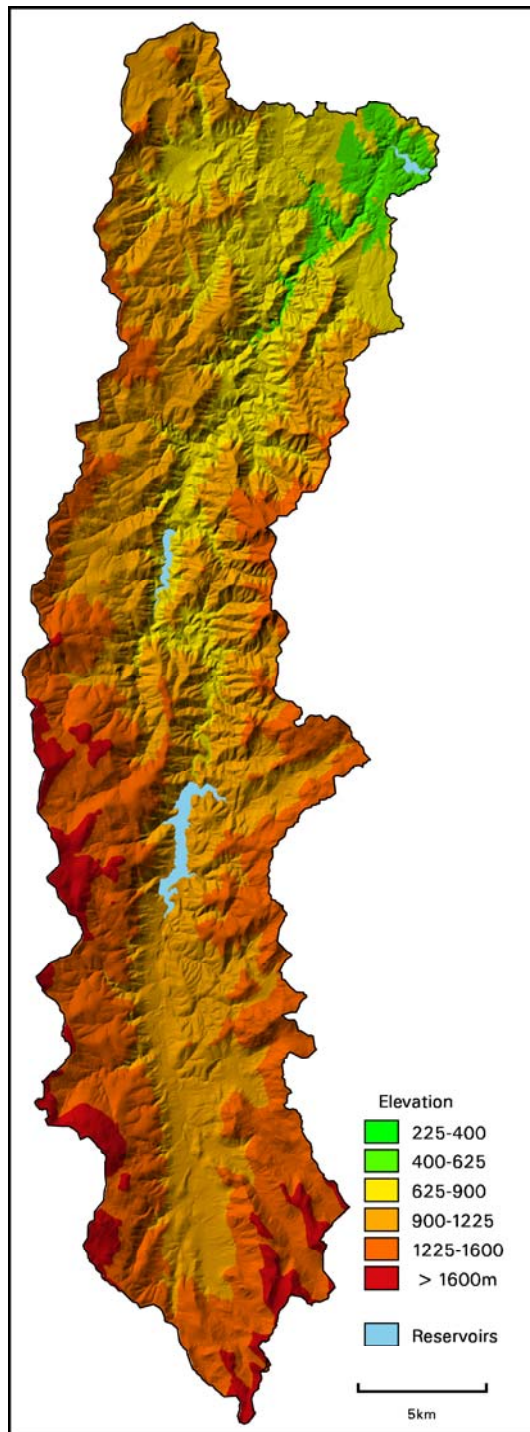


Figure 4.13. The Cotter River Catchment in southern ACT. Elevation and relief as represented by the high resolution 5m ALS DEM.

Overlapping buffer polygons for the finer scale stream networks of the Cotter catchment caused the Arc/Info BUFFER command to terminate with errors, so an alternative grid-based approach was adopted to assess the accuracy of stream placements. Each of the reference stream networks was gridded at 9-second spatial resolution, consistent with the 9-second DEM-derived network, and the proportion of the 9-second DEM-derived streamline cells that were coincident or within the immediate neighbourhood of a reference stream cell was computed. The neighbourhood of a stream cell was the default 3x3 cells used by the Arc/Info Grid FOCAL functions and thus included the adjacent grid cells in all eight directions. Coincident stream cells indicate the distance between test and reference streamlines is less than the longest axis of the grid cell, about 375m, while a neighbourhood cell is equivalent to a buffer of approximately 500 to 750m.

4.3. Results

4.3.1. Stream network

A stream network with some 955,000 segments (i.e. the section of the stream between tributary junctions) and a total stream length of approximately 2.8 million km was delineated by the drainage analysis of the 9-second DEM.

The abrupt changes in drainage density

between map sheet tiles that are a notable feature of the GEODATA TOPO-250K streamline layers are largely absent from this DEM-derived network (Figure 4.14). The total area in segment sub-catchments (Table 4.2) comprises just 76% of the computed landmass of Australia (7,711,801 km²), indicating that nearly one quarter of the continent does not drain through a stream that is shown on the 1:250,000 scale topographic map series and large enough to be resolvable at the 9-second grid resolution. More than 50% of the segments and 66 % of the total

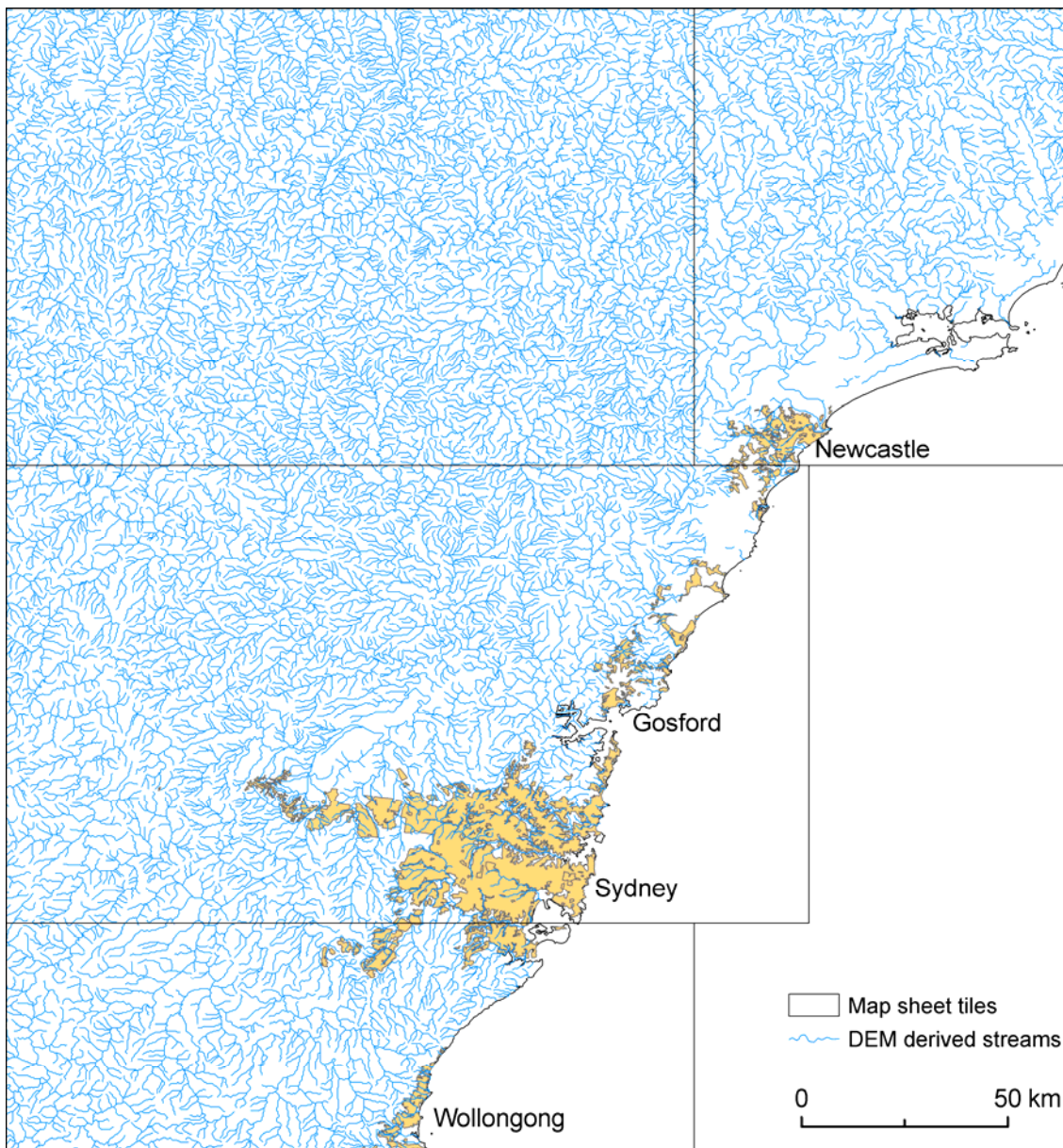


Figure 4.14 The 9-second DEM-derived stream network for the Sydney region. Compare with 3.3 on page 35

length of the network are Strahler stream order one. Larger streams, order five and above, make up just 5% of the stream segments or 2.3% of the total length.

Geometric relationships between network properties and stream order have been widely demonstrated for natural stream networks, implying regularity in network structure and supporting Horton's laws of drainage network composition (Knighton, 1998). The properties of the DEM-derived stream network show similar relationships (Figure 4.15). Even though the length of stream segments and the area that drains directly to them vary greatly (Table 4.2), they do so according to Horton's laws in a generally systematic way with network position. As predicted by the law of stream lengths, for example, there is a linear relationship between the log of the mean stream length and stream order. Similarly, conforming to the law of stream numbers, the number of streams (defined as a sequence of stream segments of the same order) varies inversely with stream order.

Table 4.2. Summary statistics for the 9-second DEM-derived stream network. Segment lengths were computed by summing the distance across the surface from the centre of one grid cell to the next in the direction of flow. In areas of higher relief these values are slightly longer than if calculated assuming a planar surface. The segment sub-catchment is the direct area draining to the stream segment.

	Segment length (km)	Segment sub-catchment area (km ²)
Minimum	0.2	0.06
Mean	2.9	6.2
Maximum	200	10,153
Total	2,804,128	5,882,408

More than 4500 bifurcations were coded into the flow direction grid to represent major anabranches and distributaries. The majority of these are associated with major lowland river systems in the Murray Darling and Lake Eyre Basins and the rivers draining to the Gulf of Carpentaria (Figure 4.16). Flow bifurcations were located on smaller streams in many other parts of the country, though less frequently. They are largely absent from regions of higher

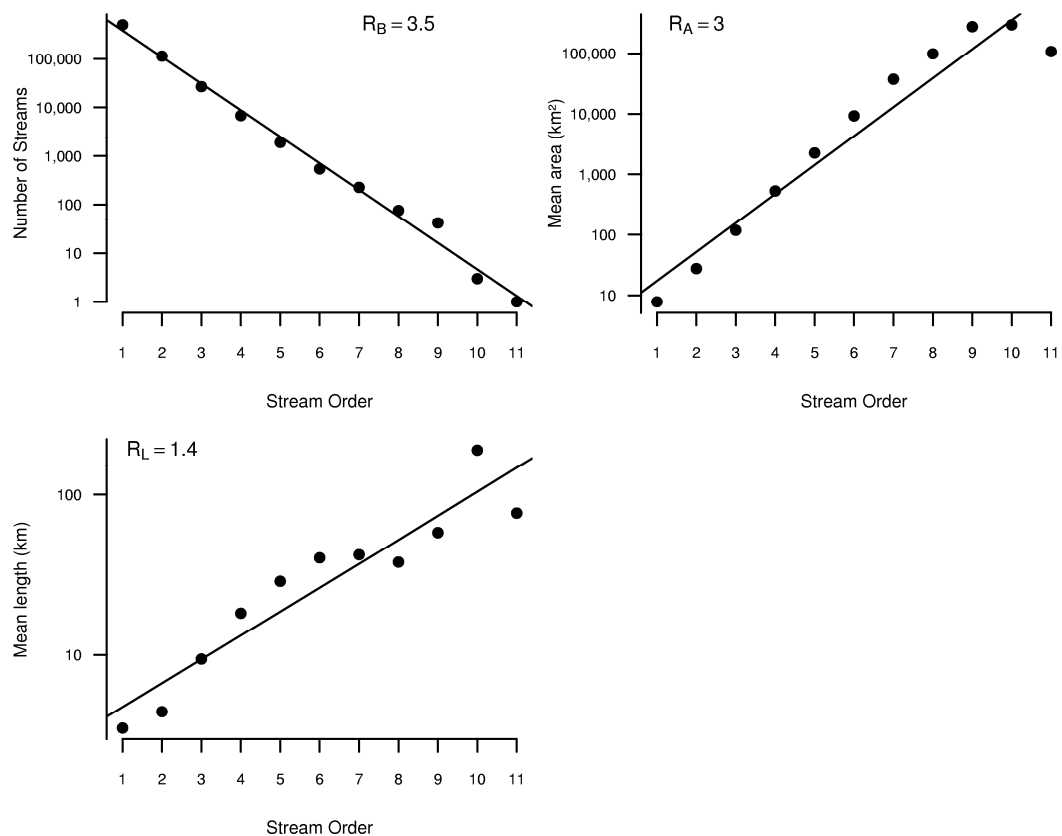


Figure 4.15 Laws of drainage network composition: properties of the derived stream network with stream order. Values for the Bifurcation Ratio (R_B), Stream Length Ratio (R_L) and Drainage Area Ratio (R_A) were derived from the slope of the linear regression.

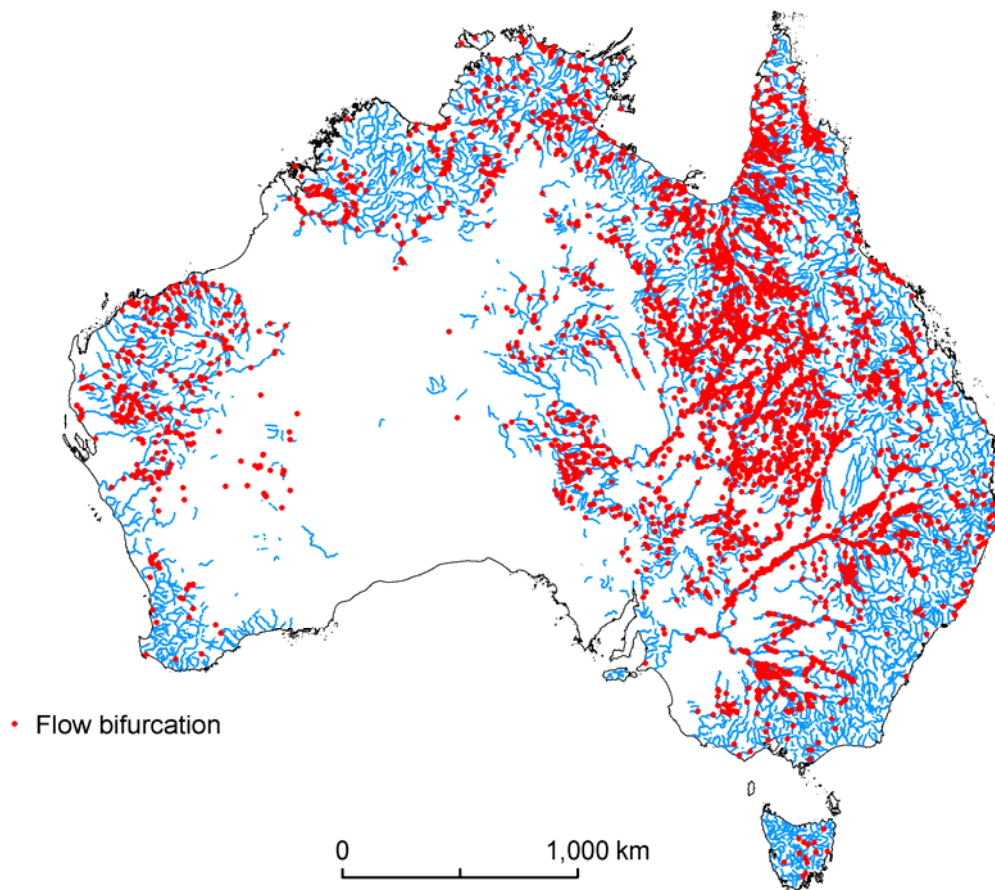


Figure 4.16 Flow bifurcations coded into the flow direction grid

relief such as along the eastern divide. A single stream path represents minor braids in the mapped streamlines where flow bifurcations were not coded.

4.3.2. Validation

The accuracy of the derived stream and catchment framework is ultimately dependent upon the ability of the 9-second DEM to reliably represent drainage structures. In this context, shape-based measures of DEM accuracy, such as the number of spurious sinks, are more meaningful than absolute measures of elevation error (Hutchinson *et al.*, 2001; Hutchinson, Stein *et al.*, 2004). The relatively few spurious sinks indicate well-connected drainage characteristics (Hutchinson, Stein *et al.*, 2004). Moreover, as an important diagnostic, sinks were instrumental in achieving a well drained DEM (Section 1.1.1).

Visual comparison of the derived stream network with digitised stream vectors from the GEODATA TOPO-250K dataset and streamline mapping on digital 1:100,000 scale topographic mapsheets at numerous locations across the country also provided qualitative evidence of drainage accuracy. More convincing evidence is afforded by the quantitative comparisons of derived catchment areas and placement of streams and drainage divides.

4.3.2.1. Catchment areas

Catchment areas computed from the DEM were found to be very close to those derived independently for the 331 stream gauging stations compiled for the NLWRA (Figure 4.17). On average, values differed by 2.5%, with more than one third of values disagreeing by less than 1%, and half by less than 1.3%. For 10 gauging stations, catchment area differed by more than 10%. The largest percentage difference in catchment area was found for gauging station number 226218, Narracan Creek at Thorpdale. The catchment provided by the NLWRA has an area of 65.8km² but that derived from the 9-second DEM only 10.4km². This difference was found to be due to an error in the DEM source data that was corrected, though too late for this analysis. The next largest difference, underestimating catchment area by 49km² or 29% (gauging station number 421084, Burrill Creek at Mickibri), arose because a small drainage area included in the catchment of the gauging station was drained to a point further downstream of the station and therefore excluded from the DEM-derived catchment. Because of the low relief, the true drainage divide could not be identified from the available 1:100,000 scale topographic mapping so it was not possible to determine which of the catchment boundaries for this station was more correct.

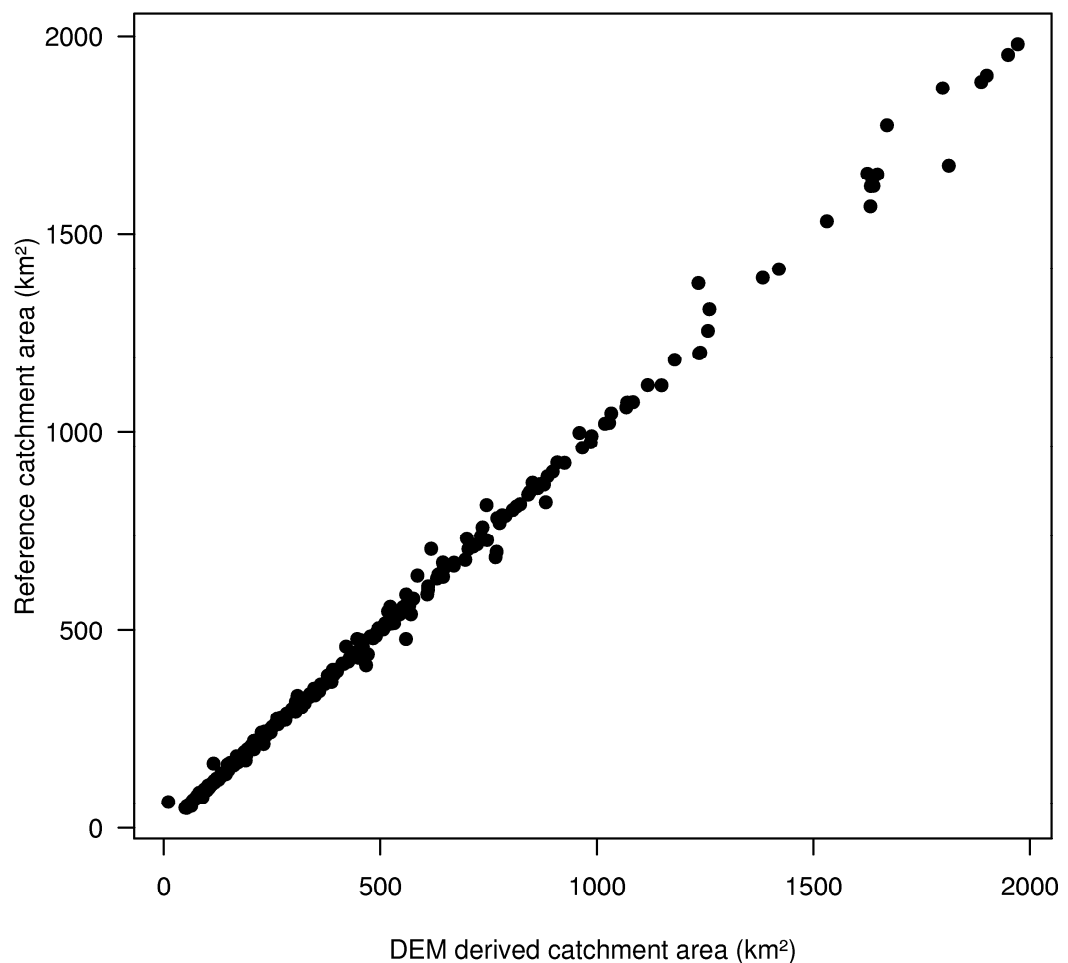


Figure 4.17. Comparison of DEM-derived and reference areas for catchments of the NLWRA extended flow series gauging stations.

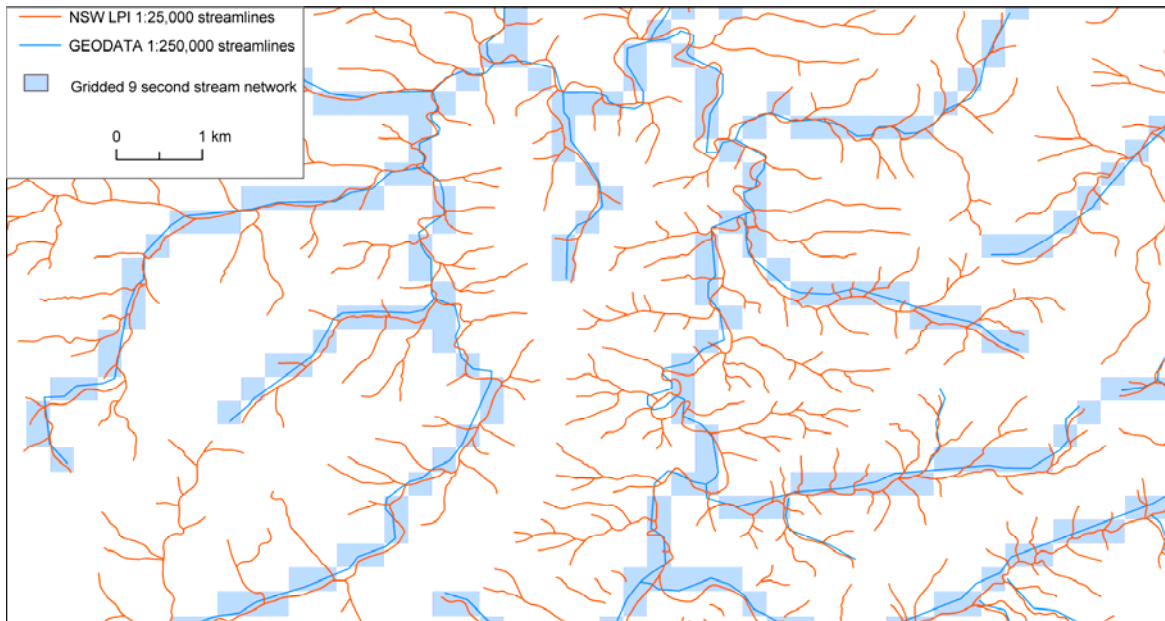


Figure 4.18 DEM-derived streamlines with mapped streamlines from the 1:250,000 scale GEODATA TOPO 250K database and NSW 1:25,000 scale topographic mapping.

4.3.2.2. Stream locations

Almost all (98%) of the grid cells in the DEM-derived stream network are coincident with a streamline on the GEODATA 1:250,000 scale topographic maps, gridded at the same 9-second spatial resolution (Table 4.3). Nearly half of the remaining cells in the DEM-derived network are adjacent to a grid cell traversed by a mapped streamline. The generalization of the streamline source data by the ANUDEM gridding program, in particular the removal of excessive kinks, explains many of these small discrepancies in stream locations (Figure 4.18). Many of the cells not matching the gridded representation of the GEODATA streams will fall in the short gaps (<5km) in the GEODATA streams that were joined in the DEM-derived network.

The 9-second DEM-derived stream network also compares well with finer scale streamline mapping for the Cotter River catchment when compared at the same 9-second scale (Table 4.4). Clearly, however, at finer scales the 9-second DEM-derived network is more generalized (Figure 4.18) and predictably, substantially underestimates the extent of the stream network (Table 4.5). The density of streamline mapping on the ACT 1:10,000 scale map series is highly variable across the catchment and, surprisingly, in some areas it is less extensive than the GEODATA 1:250,000 scale mapping. This fact probably explains the poorer performance of the derived stream network against this layer.

Table 4.3. Coincidence of the 9-second DEM-derived stream network and gridded GEODATA TOPO 250K 1:250,000 scale streamlines

Status	Number of grid cells (% total stream cells)
No match	112,626 (1.3%)
Stream cells coincident	8,690,012 (97.6%)
Adjacent cell traversed by mapped stream	102,634 (1.2%)

Table 4.4 Coincidence of the 9-second DEM-derived stream network and gridded reference streams in the Cotter River catchment

	Number of grid cells (% total stream cells)		
	NSW 1:25,000 scale topographic mapping	ACT 1:10,000 scale topographic mapping	Stream network derived from 5m ALS DEM ₁
No match	1 (0%)	42 (4%)	0
Stream cells co-incident	1158 (96%)	1075 (89%)	1168 (97%)
Adjacent cell traversed by reference stream	47 (4%)	89 (7%)	38 (3%)

1. Using a contributing area threshold of 2.5ha

Table 4.5. Cotter River catchment total stream length

Source	Total length of stream (km)
9-second DEM-derived	365
GEODATA TOPO-250K	375
NSW 1:25,000 scale topographic map series	1393
ACT 1:10,000 scale topographic map series	1023
5m ALS DEM-derived (2.5ha contributing area threshold)	1734

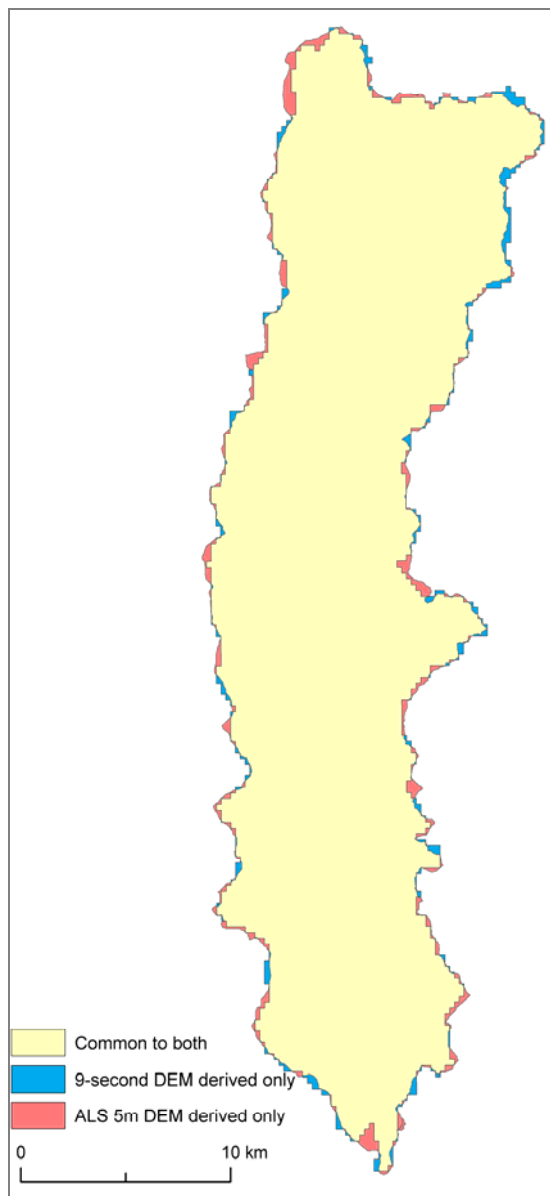


Figure 4.19. Comparison of the Cotter River catchment boundaries delineated from the 9-second DEM and 5m ALS DEM

4.3.2.3. Drainage divides

Drainage divides were found to be similarly accurate with almost all of the DEM-derived catchment boundaries within 500m of the reference catchment (Table 4.6). Some errors in the NLWRA catchment boundaries were identified when a sample of the largest departures (identified by visual inspection) from the boundaries supplied for the NLWRA series of gauging stations were checked against 1:100,000 topographic maps. For the vast majority of its length, the catchment boundaries of the Cotter River catchment delineated from the three DEMs (9 second, 20m and 5m) were closely aligned (Table 4.6). Nevertheless, there are obvious differences due to scale that are reflected in the figures for the total length of the catchment boundary: 186km from the 9 second DEM; 206km from the 1:25,000 scale 20m DEM and 211km from the 5m ALS DEM. At finer scales, the catchment divide is significantly more detailed and hence longer and conforms more closely to the actual divide. The area of the catchment delineated

from the 5m ALS DEM (48,112ha) is also slightly larger than the area derived from the 9 second DEM (47,660ha) but more than 97% of this area was common between the two (Figure 4.19). The differences due to scale are most evident when the boundaries of individual sub-catchments are compared (Figure 4.20).

Table 4.6. Proportion of the 9-second DEM-derived catchment boundary length within the buffer distance of the reference catchment boundary.

Buffer size (m)	Reference catchments		
	NLWRA Extended flow series	Cotter catchment (1:25K 20m DEM)	Cotter catchment (ALS 5m)
250	90.6%	95.7%	95.4%
500	97.3%	99.9%	99.9%
1000	98.2%	100%	100%

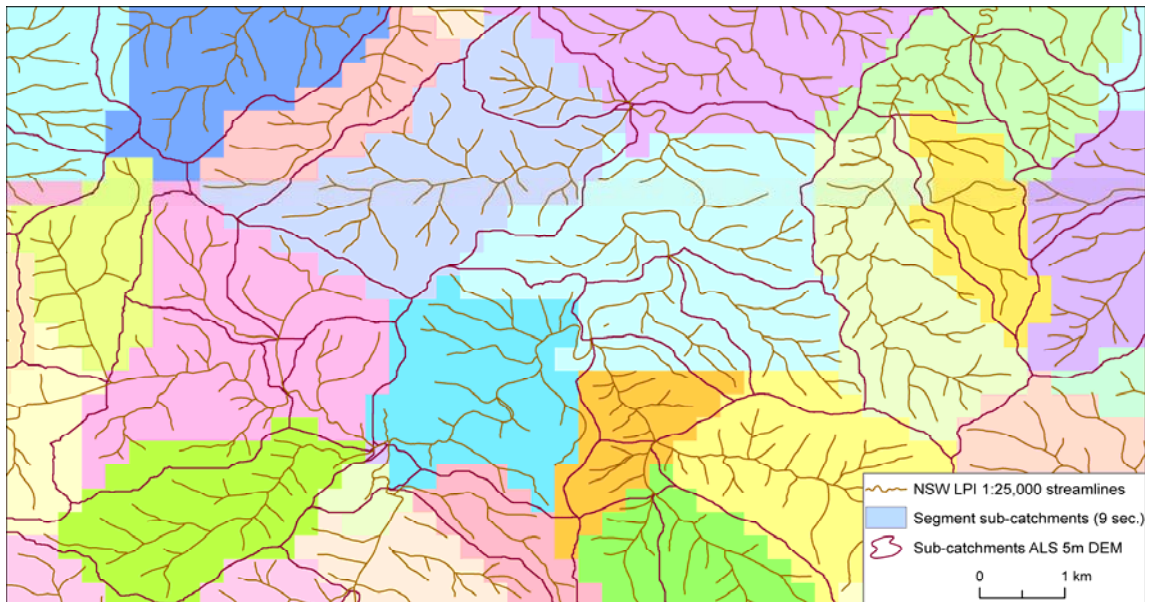


Figure 4.20 Comparison of Cotter River segment sub-catchment boundaries. Sub-catchments delineated from the ALS 5m DEM were seeded with a contributing area threshold of 1.25km^2

4.4. Discussion

4.4.1. Drainage analysis procedures

There is growing recognition of the value of consistent, continent-wide delineation of hydrologic units (Bertolo, 2000; Franken *et al.*, 2001; U.S. Geological Survey, 2001). Although developed for Australian drainage systems, the new drainage analysis procedures are generic and applicable anywhere that basic topographic mapping is available. They overcome many of the limitations of existing methods for continental scale application, effectively accounting for distributary and uncoordinated drainage systems and accurately delineating stream networks, even in regions of low relief.

These procedures recognise DEM development as a fundamental step in the drainage analysis. The accuracy of the drainage analysis products depends on the ability of the DEM to correctly represent surface drainage characteristics and is thus reliant on selection of appropriate DEM data sources and interpolation methods, and careful attention to error correction. Additional investment in DEM development, moreover, largely removes the need for DEM pre-processing routines such as stream burning and filling. Such pre-processing can be a significant and time-consuming undertaking. The Elevation Derivatives for National Applications (EDNA) project in the United States (Verdin and Greenlee, 2003), for example, oversees a major cooperative program to identify and correct errors and inadequacies in the national DEM (Gesch *et al.*, 2002) to improve hydrological derivatives (Kost *et al.*, n.d.). The production of a high quality

DEM has the added benefit of supporting other applications requiring terrain derivatives (e.g. landscape classification, see Chapter 7).

Setting appropriate thresholds for channel initiation is always challenging, but especially so at continental scales (Section 3.2.1.1). The solution developed here makes use of commonly available information in the form of cartographic streamlines. It does not attempt to define the true extent of the stream network, accepting that it cannot be reasonably determined from continental scale DEMs. Furthermore, the interface between hillslope and channel is highly dynamic (Kim and Lee, 2004). “Growing” the channel network from designated “start” cells is not a new idea (see for example review in Bertolo, 2000) but, surprisingly, the mapped channel heads do not appear to have been used to initiate the network.

The new drainage analysis procedures represent the major features of the drainage network without attempting to resolve the complex patterns of braided and anastomosed systems. For continental scale conservation planning it is not necessary (nor feasible) to model the complex fluvial and erosional processes (Makaske, 2001; Fagan and Nanson, 2004) responsible for the evolution of multi-channel planforms. Representing only major flow bifurcations avoids the problems of looping flow paths and other anomalies that may be encountered when multiple flow directions are included at every bifurcating node identified in the mapped streamlines (Stein *et al.*, 2002). The number of flow bifurcations incorporated into the stream network is controlled by the value of the area threshold used to delineate sub-catchments. A reasonable threshold value was determined by trial and error from visual inspection of the resulting stream networks for selected test areas. Recognising flow bifurcations only where the branching stream arcs flowed to different terminal cells (inland sinks or coastal outlets) might be a more robust approach but would exclude distributaries within a drainage basin (e.g. the Cuttaburra/Kulkyne Creek system that links the Paroo and Warrego Rivers, in the Murray-Darling Basin). The specification and operation of the criteria used to identify major bifurcations would benefit from further research.

Stream segments are bounded by tributary confluences. Ideally, stream segments are also broken at natural discontinuities such as major breaks in slope (Frissell *et al.*, 1986) but developing a reliable method was not feasible for this study. A simple algorithm that breaks segments at grid cells overlayed by a cliff line and with a large drop in elevation from one cell to the next in the flow direction was tested, but requires further development. Many of the problems encountered could be attributed to the relatively coarse grid cell resolution so, for example, cliffs that were close to a confluence were attributed to the wrong stream segment and segments were erroneously divided where cliff lines ran close to, but parallel with, the stream (i.e. they delineated a gorge).

4.4.2.Drainage analysis of the Australian DEM

The drainage analysis products that are the outcome of the application of the new procedures reflect the natural surface drainage characteristics of the Australian continent. Application of objective, systematically applied criteria produced a consistent representation of drainage networks without the confounding influences of variable cartographic interpretation. The geomorphometric properties of the derived stream network generally conform to the laws of drainage basin geometry usually ascribed to natural stream networks, supporting a contention that the intrinsic organization of river systems is preserved in the DEM-derived network (Vörösmarty *et al.*, 2000).

Values for the Horton ratios typically fall within the range 3 to 5, 1.5 to 3 and 3 to 6 for the bifurcation (R_B), stream length (R_L) and drainage area (R_A) ratios respectively (Kirchner, 1993). The ratio values for the DEM-derived stream network ($R_B = 3.5$, $R_L = 1.4$, $R_A = 3$) are thus towards the lower end or in the case of R_L just outside of the usual range. Lower values are largely explained by the incorporation of uncoordinated and distributary drainage structures. Mean stream lengths will be lower than expected as a result of the breaks inserted into the DEM-derived network that correspond with gaps in the mapped streamline network and difficulties assigning stream orders to distributary channel networks. The tests used to check for bifurcating channels re-joining, based on the elevation of the channel source, failed in certain circumstances, causing stream orders to be falsely incremented. Downstream of the confluence between an anabranch and a tributary originating from a higher source, the anabranch will be attributed as being derived from a new source (i.e. that of the tributary). As a consequence, stream order will be incremented if this anabranch rejoins the main stream further downstream, a situation demonstrated by the Flinders River in the Gulf of Carpentaria, the only river of order eleven. Inflated stream orders assigned to some distributary rivers will also cause the mean drainage area to be lower than is characteristic for a stream order. The single continent-wide value of the Horton ratios reported here may also be masking considerable variation between regions or drainage basins that would be usefully explored in the future.

4.4.3.Validation

The accuracy assessment undertaken in this study indicates that the new stream and catchment delineations will provide a reliable basis for continental scale conservation planning. The assessment corroborates the view of Hutchinson *et al.* (2000) that, at a nominal scale of about 1:250,000, the true surface drainage structure of the Australian continent is reliably determined from the national 9 second DEM. This scale is consistent with that recommended by the scientific guidelines developed for the National Reserve System Program of Australia

(Commonwealth of Australia, 1999) and is appropriate for regional to continental scale planning.

The buffer analysis and visual inspection of stream placements demonstrates the ability of the ANUDEM program to accurately represent drainage structures in the DEM, even in areas of low relief. Therefore, it avoids the need for stream burning or other DEM pre-processing algorithms. In most cases, divergence from the mapped streamlines can be attributed to the generalization of the input source data to the grid resolution by the ANUDEM program. Version 2 of the GEODATA TOPO –250K database (Geoscience Australia, 2003c) incorporates recent revisions to streamline mapping and potentially, therefore, locates streamlines more accurately. The version 2 database is available for free download. However, comprehensive coverage was only available from early in 2004. This was too late for this project. Disappointingly, the streamline vectors in version 2 do not retain the correct directions established for the version 1 streamline vectors. Furthermore, the connectors, added to bridge natural gaps in the drainage network, were poorly located, often taking little account of the underlying topography. In their current form then, the GEODATA version 2 streams are unsuitable for drainage analysis and DEM development.

The results of the buffer analysis compare favourably with those reported by Vogt, Colombo, Paracchini *et al.* (2003) for a pan-European stream and catchment dataset derived from a 250m resolution DEM. Despite the application of adaptive drainage enforcement algorithms (Soille *et al.*, 2003) to improve the accuracy of the DEM flow paths in flat areas, the European stream network fell within a 250m buffer of the 1:50,000 scale reference streams for only 50% of its length, or within a 500m buffer for 87% of its length. In comparison, for all but the ACT 1:10,000 scale stream mapping, more than 96% of the grid cells of the stream network derived in this study were coincident with the rasterised reference stream network, indicating their locations differed by not more than about 375m. The highly variable mapping of drainage extent on the ACT 1:10,000 scale map series confounded comparison with the derived stream network. The accuracy assessment may, therefore, have been improved by considering only selected streamlines common to both the derived and reference datasets.

The buffer analysis of catchment boundaries showed that the drainage analysis of the Australian DEM also more accurately delineated catchment boundaries than the comparable European analysis. Less than 85% of the length of the derived catchment boundaries in the European dataset fell within a 1000m buffer around the reference catchment boundaries (Vogt, Colombo, Paracchini *et al.*, 2003). In contrast, in this study more than 95% of the length of the derived catchment boundaries was within 500m of the reference set. This figure was almost 100% for the higher resolution reference data sets defining the Cotter Catchment boundary.

Summary statistics can, however, hide localized elevation errors that potentially have a very large influence on derived drainage networks and catchment areas (Jenson, 1991; Walker and

Willgoose, 1999). While the likelihood of such effects is substantially reduced by the comprehensive error correction procedures that were integral to DEM development and by the use of flow directions directly coded from the mapped streamlines, such errors need to be considered as a potential source of uncertainty in conservation planning applications based on the derived drainage network.

4.5. Conclusions

The new drainage analysis procedures developed in this study provide effective solutions to the challenges of delineating a continent-wide stream network that accommodates the range of observed drainage structures. They recognise the limitations of continental scale data and therefore do not attempt to model the physical processes of stream network development. The data required to apply these procedures are commonly available from topographic mapping so they could be readily applied to other continents. Still, opportunities to improve the algorithms were identified and these could provide the focus for future research.

Chapter 5. A nested catchment framework

Chapter 4 describes an original method of drainage analysis and demonstrates its application to delineate a continent wide stream network. However, as Chapter 2 (Section 2.3) explains, the stream system encompasses a nested hierarchy of hydrologic units. Accordingly, this chapter describes the development of a multi-scale nested catchment framework, including a major revision of the drainage basins of Australia and the first Australia-wide application of the Pfafstetter system, a particularly efficient method for delineating and coding hierarchically nested catchments that has been adopted internationally (Section 3.2.2).

An account of the methods employed to derive the new nested catchment framework is presented in Section 5.1. It includes a description of the procedures that were developed to delineate drainage basins and to aggregate small closed basins. The Pfafstetter system is explained and specific details of its Australian implementation clarified. The results are presented in the following section (Section 5.2). These focus firstly on the properties of the revised drainage basins and their comparison with the widely adopted AWRC framework, and secondly, on the application of the Pfafstetter system. The final section of this chapter (Section 5.3) discusses the suitability of the catchment framework for conservation planning applications and the utility of the Pfafstetter system.

5.1. Methods

5.1.1. Delineating drainage basins

In preparation for application of the Pfafstetter system, drainage basins were delineated and their areas computed. Basins, being the entire catchment that drains to a river mouth or terminal lake, were seeded by inland grid cells with undefined flow direction (i.e. sinks) or on the coast, with a cell directed to a no data value cell (i.e. ocean). The catchment of each uniquely numbered basin seed was defined using a modified version of the FORTRAN90 flow accumulation program (Section 4.2.1.4). Rather than accumulating cell values downstream, the program tracks the flow paths upstream to their source, assigning to each cell the identifier of the last seed encountered. A cell upstream of a bifurcation might be reached from more than one seed. These conflicting assignments were flagged and later reassigned a single numeric identifier, arbitrarily, that of the seed with the lowest identifier among all connected cells.

The area of each basin was computed by summing the area of individual cells, calculated from the geographic coordinates using trigonometric relationships and the radius of the earth (6,370.997km). For comparison with other continental drainage analyses and to highlight broad

scale topographic structure, the internally draining basins were iteratively merged with a lower neighbour using the method described below (Section 5.1.2), applying sink depth thresholds in the range 10 to 100m to produce various levels of aggregation. Following previous convention (Hutchinson and Dowling, 1991), basins were named according to either the largest stream or the waterbody feature at their lowest point. Names were automatically assigned from the drainage and waterbody layers from the GEODATA TOPO-250K database version 1 (AUSLIG, 1992b).

5.1.2. Aggregating small basins

An ArcInfo AML routine was written to aggregate small closed (endorheic or internally draining) basins to facilitate investigation of large-scale drainage structures and allocation of Pfafstetter codes. The routine, adapted from one developed for an earlier project (Hutchinson, Stein *et al.*, 2000), is modelled on the sink-filling procedures described in the Arc/Info manuals (ESRI, 2004). It iteratively aggregates small closed basins by assigning them the identifier of the neighbouring catchment via the saddle point, the lowest cell on the boundary of the sink catchment, until the desired level of aggregation is reached. Unlike the Arc/Info procedures, however, it does not alter the flow direction or elevation values and considers grid cells to be on the catchment perimeter using 8-way connectedness, not just the four cardinal directions. Thus the aggregated catchments are treated as one for all uses that rely only on the zone identifier (e.g. the zonal summary functions in Arc/Info Grid) but retain original surface flow pathways for all flow routing applications.

Closed basins are aggregated in descending order according to the elevation of their lowest (sink) point. Saddle point cells are perimeter cells of closed basins with an elevation that is within a small tolerance of the lowest perimeter cell. The identifier of the lowest catchment adjacent to a saddle point is assigned to the closed basin and the perimeter cells redefined (Figure 5.1). The elevation tolerance improves the opportunities for aggregation because the absolute lowest perimeter cell may not always be adjacent to a lower neighbour, especially in small basins. The threshold value used (0.2m) is well within quoted errors for the DEM (Hutchinson *et al.*, 2001). A sink depth threshold controls the extent of aggregation. Basins are only aggregated where the height of the saddle point above the lowest point in the sub-catchment (i.e. the sink depth) is below the specified threshold.

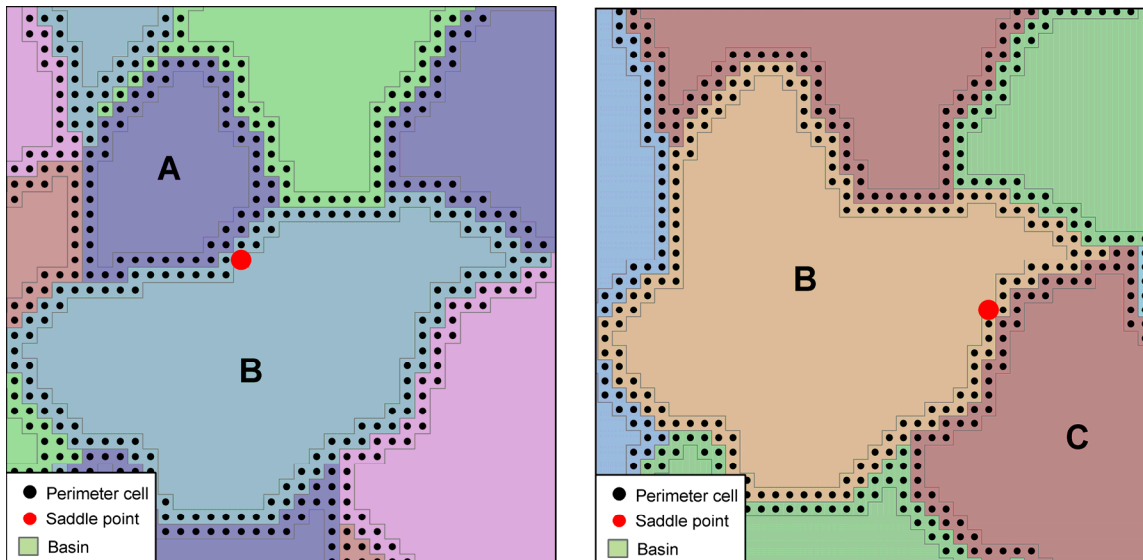


Figure 5.1 Aggregating small closed basins for assignment of Pfafstetter codes and identification of broad scale topographic structure. a) Step 1: identify the saddle point of the basin with the highest sink value (basin A). In this example, no other grid cell has an elevation value within the range of the minimum perimeter value $\pm$ the elevation tolerance; b) Step 2: basin A is assigned the identifier of its neighbouring (lower) basin (basin B) via the saddle point. The perimeter cells of basin B are redefined and its saddle point identified. Basin B would next be merged with basin C.

5.1.3. Implementing the Pfafstetter system

The Pfafstetter system (Verdin and Verdin, 1999) is a global reference scheme for sub-dividing and coding drainage basins on the basis of the topology of the drainage network and the size of the drainage area. The system delineates and numerically codes topographically defined hydrological units that it labels “basins” and “inter-basins”. A basin is the area drained by a tributary. The area directly drained by the reach of the main stem lying between two tributaries is labelled an inter-basin. The Pfafstetter system defines the main stem as the one draining the larger area, even where this may contradict usual naming conventions. The coding scheme uses only the digits 0 to 9 for each level of sub-division. Basins and inter-basins are repeatedly sub-divided into four major tributary sub-basins, coded with the even digits, and five inter-basins, coded with the odd digits, as many times as supported by the DEM. A closed basin, that is the area draining to an internal sink, is assigned a code of zero.

The Pfafstetter coding scheme restricts sub-division of a single hydrological unit to no more than nine sub-units. When applied to a continent with many thousands of drainage basins, an additional procedure for aggregating basins is needed to keep the number of units within these limits. Verdin and Verdin’s (1999) method is to treat the adjacent groups of small coastal draining basins that lie between the four largest drainage basins as coastal inter-basins. The first (Level 1) sub-division of a continent thus delineates the largest closed basin and the four largest of the coastal basins and allocates the remaining coastal areas to coastal inter-basins.

Subsequent levels sub-divide basins according to the Pfafstetter system and coastal inter-basins by applying the continental Level 1 procedure within each coastal inter-basin.

This scheme was applied to divide the Australian continent, delineating a hierarchical, spatially nested catchment reference system. First the four largest coastal basins were identified and numbered with the even digits 2, 4, 6 and 8 in a clockwise direction from the most northern tip of Australia on Cape York Peninsula. Next the largest internal closed basin was coded with a value of 0. The smaller coastal draining basins lying between the 4 largest basins already coded were grouped and these coastal inter-basins coded with the odd digits 1, 3, 5, 7 and 9 in a clockwise direction. Small basins on islands were allocated to the nearest mainland coastal basin. To complete the level 1 sub-division the unallocated internally draining basins were iteratively assigned the code of the adjacent basin via the lowest saddle point as described above (Section 5.1.2) disregarding sink depth, until all basins were allocated a Pfafstetter code (Figure 5.2).

This procedure was repeated for the level 2 sub-division. Each of the Level 1 coastal inter-basins was sub-divided into the four largest basins, numbered again with the even digits and the intervening inter-basin areas with the odd digits (Figure 5.3). The Level 1 basins, including the closed basin labelled 0, were internally sub-divided into tributary “basins” and main stem inter-basins. In this context, Verdin and Verdin’s (1999) basins are equivalent to the catchments in

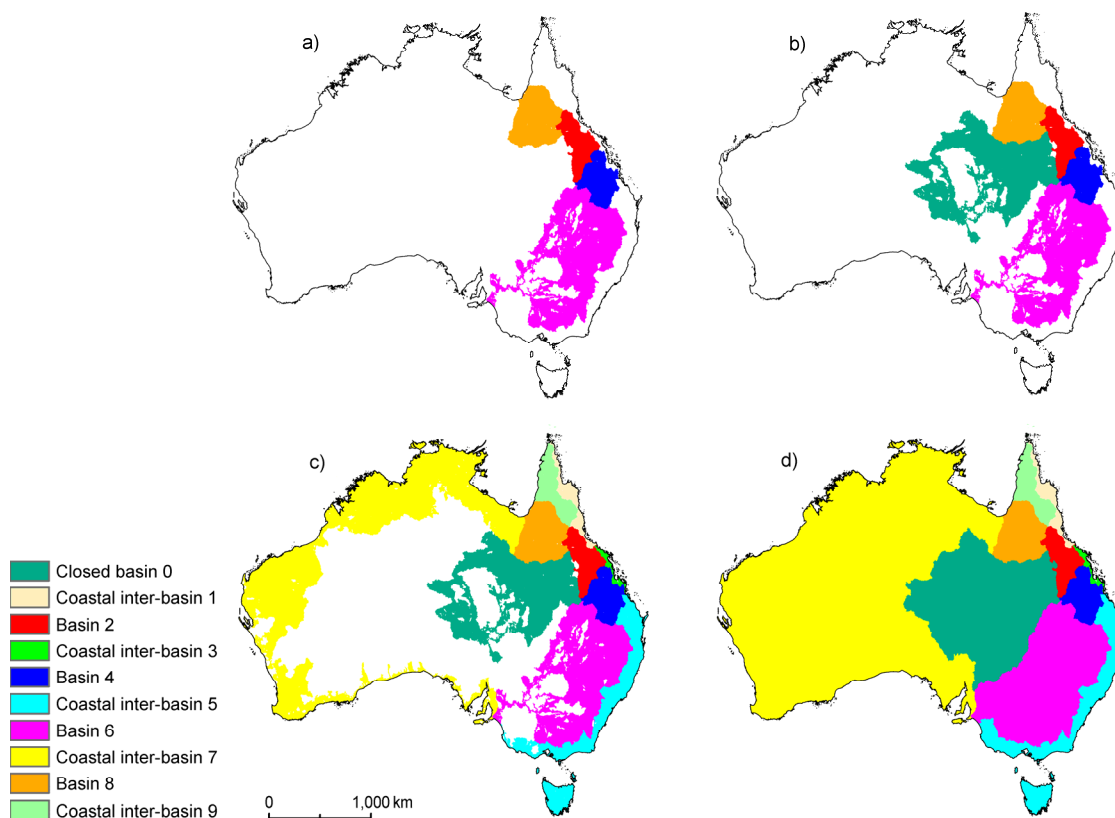


Figure 5.2 Pfafstetter Level 1 sub-division of the Australia continent a) identifying and coding the four largest coastal draining basins; b) the largest closed basin; c) delineating coastal inter-basins between the largest coastal draining basins and d) assigning a Pfafstetter code to all remaining closed basins by iteratively allocating them to a lower neighbour.

Table 2.1 on page 27.

To differentiate tributary and main stem streams it was necessary to use the connected version of the DEM-derived stream network. This is the network prior to deletion of any of the sections corresponding with gaps in the streamline mapping or overlaying major lakes (Section 4.2.2.3).

At each confluence the main stem was identified as the stream draining the larger area.

Upstream contributing area was computed by accumulating the area of all grid cells upstream to the centre of the stream segment pour-point cell. Upstream areas were not divided at anabranches. The four largest tributaries off this main stem were numbered with the even digits 2, 4, 6 and 8 in sequence from the catchment outlet. The inter-basin areas draining directly to the main stem between tributary junctions were assigned the odd digits 1 to 9, again in a sequence upstream from the catchment outlet (Figure 5.4a). The largest internal closed basin

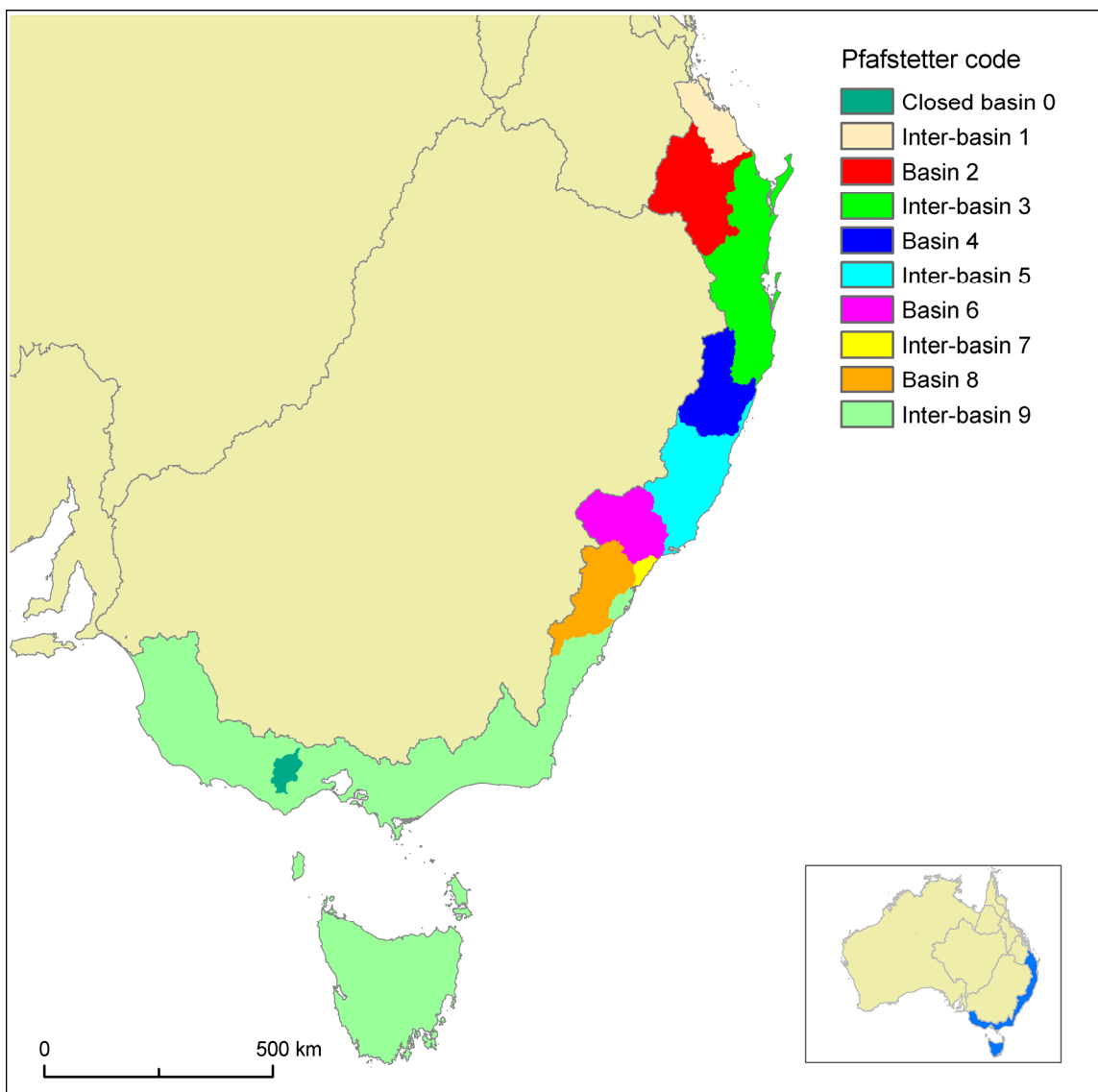


Figure 5.3 Pfafstetter sub-division of a coastal inter-basin (Level 1 inter-basin 5).

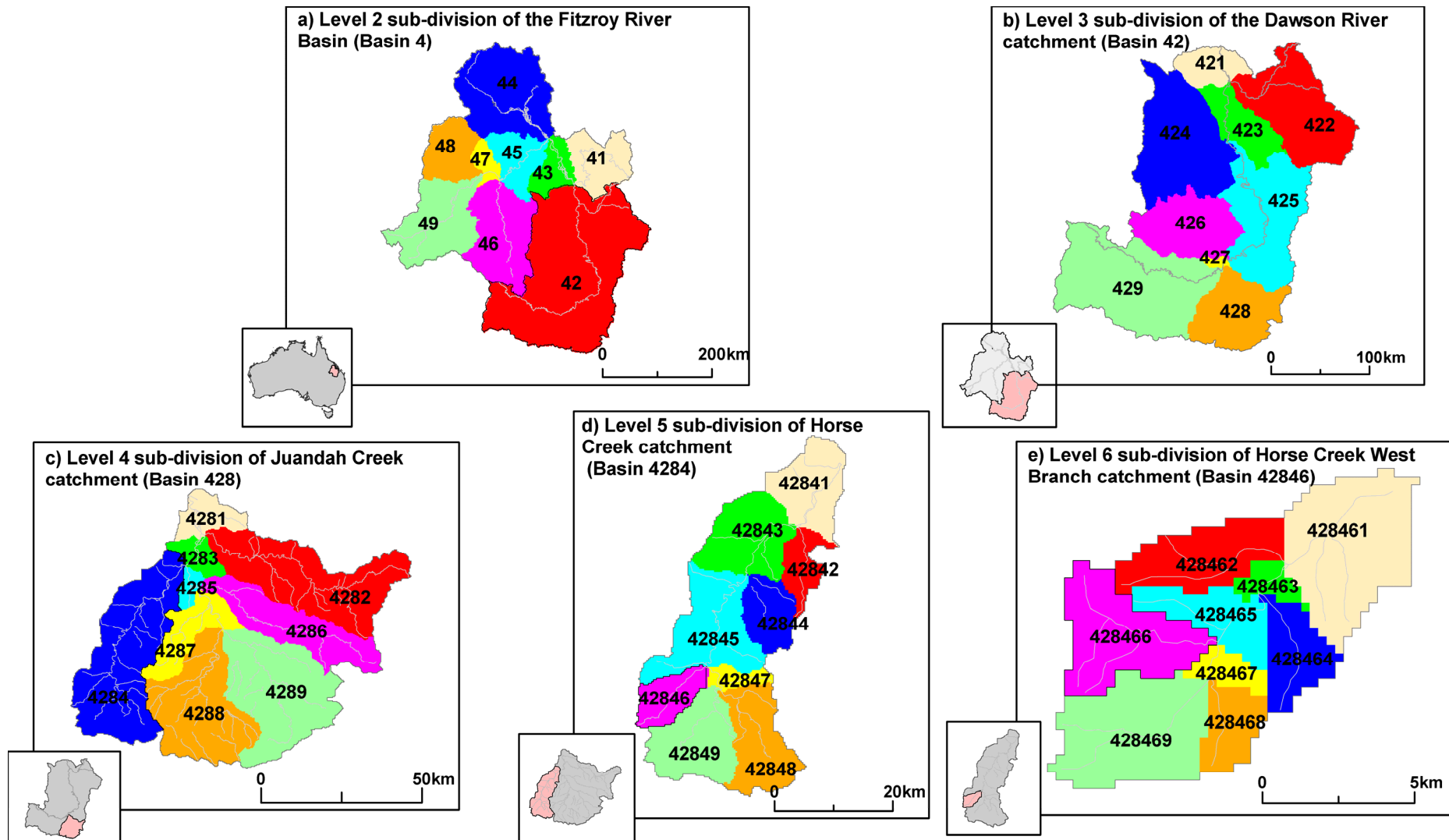


Figure 5.4 Pfafstetter sub-division of a basin: Fitzroy River basin (Level 1 basin 4).

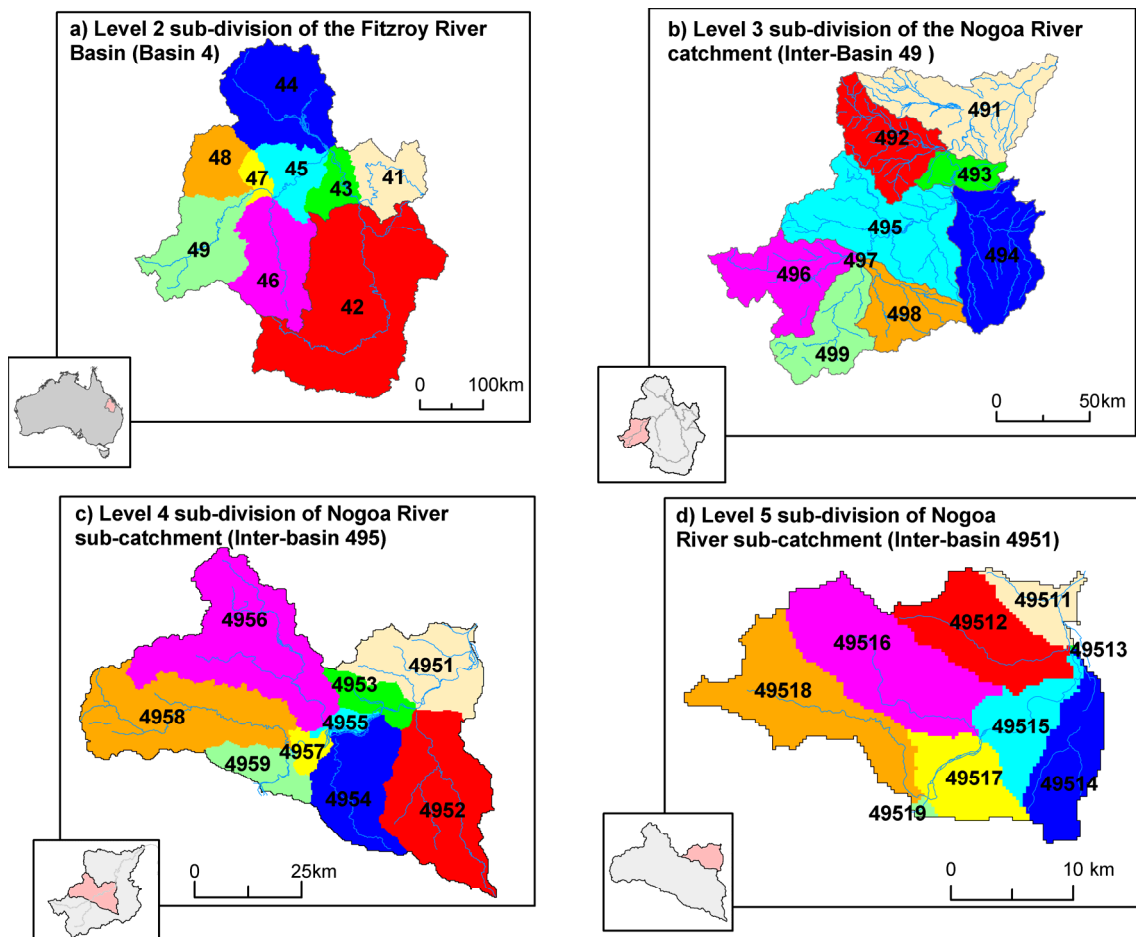


Figure 5.5 Pfafstetter sub-division of an inter-basin: the Nogoa River sub-catchment (Level 2 inter-basin 49) of the Fitzroy River basin (Level 1 basin 4).

within each Level 1 basin and inter-basin was assigned the code 0 and as before, remaining closed basins allocated the Pfafstetter code of a lower neighbour. This completed the Level 2 Pfafstetter sub-division.

The procedure can be repeated as many times as supported by the resolution of the DEM to produce increasingly finer divisions of basins (Figure 5.4) and inter-basins (Figure 5.5). Verdin and Verdin (1999) proposed stopping basin sub-division when four tributaries can no longer be found. For particular channel network configurations, however, this rule could leave large tributary basins undivided. Hence, sub-division of basins was allowed as long as at least one tributary was identified. When no further sub-divisions were possible a code of 0 was applied to the catchment unit for this and subsequent levels.

Procedures were programmed to ensure that the deltaic distributary systems of channels that characterise some tributary-main stem confluences were treated as a single unit even where more than one tributary channel connected to the main stem (Figure 5.6). A single identifier was assigned to all connected tributaries by delineating their catchment using the FORTRAN programs developed in this study, seeded by the cell immediately upstream of the confluence with the main stem. Thus all tributary channels within the same tributary catchment were treated as a single tributary for Pfafstetter code allocation. In addition, any channel joining a main stem

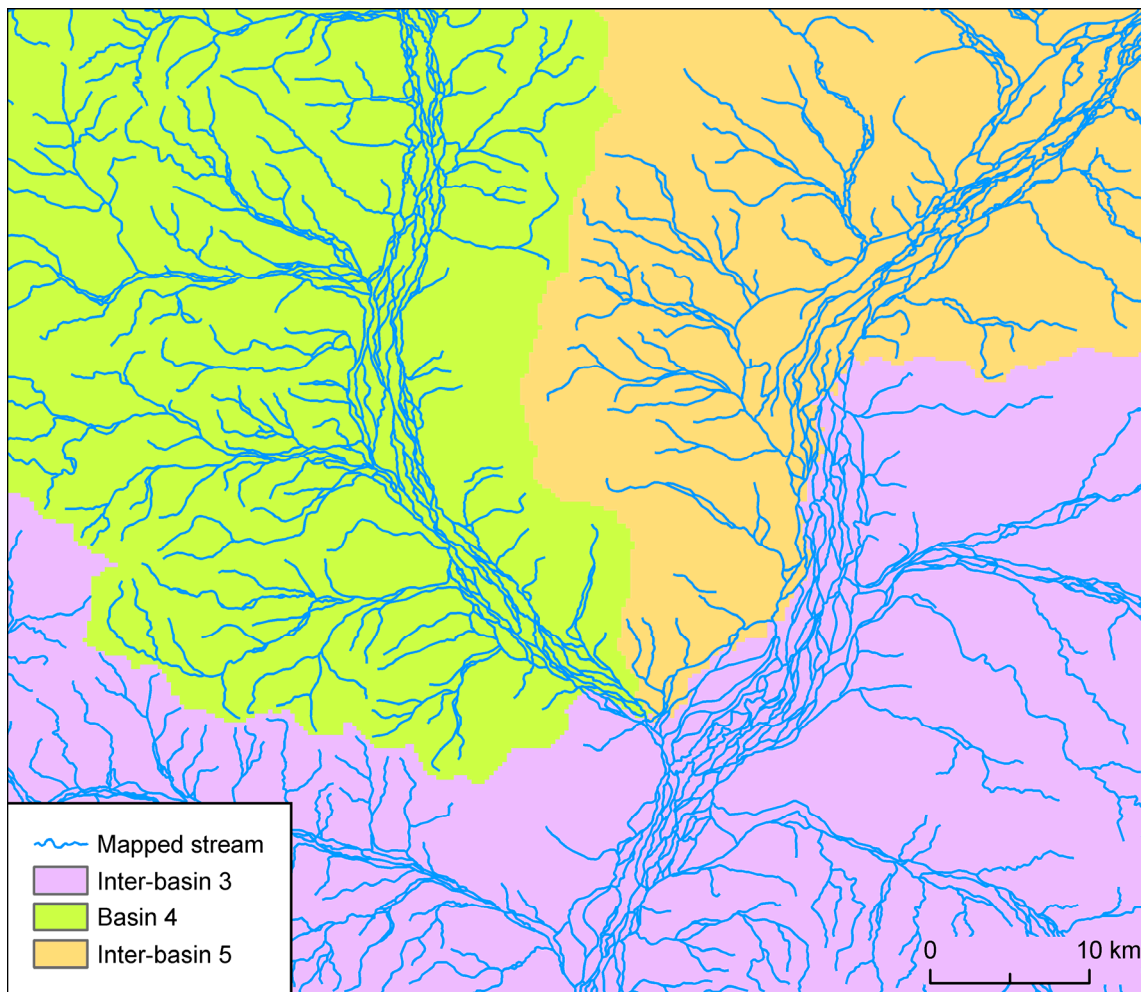


Figure 5.6 Treating the multiple channels of distributary channel systems as a single tributary unit: Vergemont Creek and Thomson River confluence, Lake Eyre Basin.

that was located entirely within a valley bottom flat (Appendix 1) was considered more likely to be an undetected floodplain channel and therefore included with the main stem inter-basin sub-catchment.

The increasingly finer division of basins and inter-basins was continued until Level 9 Pfafstetter units were defined. By this level, most basins had been sub-divided as far as they could for the given stream network configuration. Pfafstetter codes were formed by combining the single digit codes from each of the higher level sub-divisions. For example, the Level 3 code 682 indicates a tributary of basin 68 which itself is a tributary of basin number 6.

Differentiating tributary and main stem segments according to upstream catchment area sometimes contradicts usual conventions. The Murray River, for example, drains a smaller area than the Darling River at their confluence but the Murray River carries the greater flow and lends its name to the river downstream of the confluence. To address this concern, an alternative nine level Pfafstetter sub-division was derived using a simple, modelled catchment water balance as an indicator of annual mean flow volume in place of catchment area. Section 7.2.2.1 provides details of the procedures used to derive the water balance.

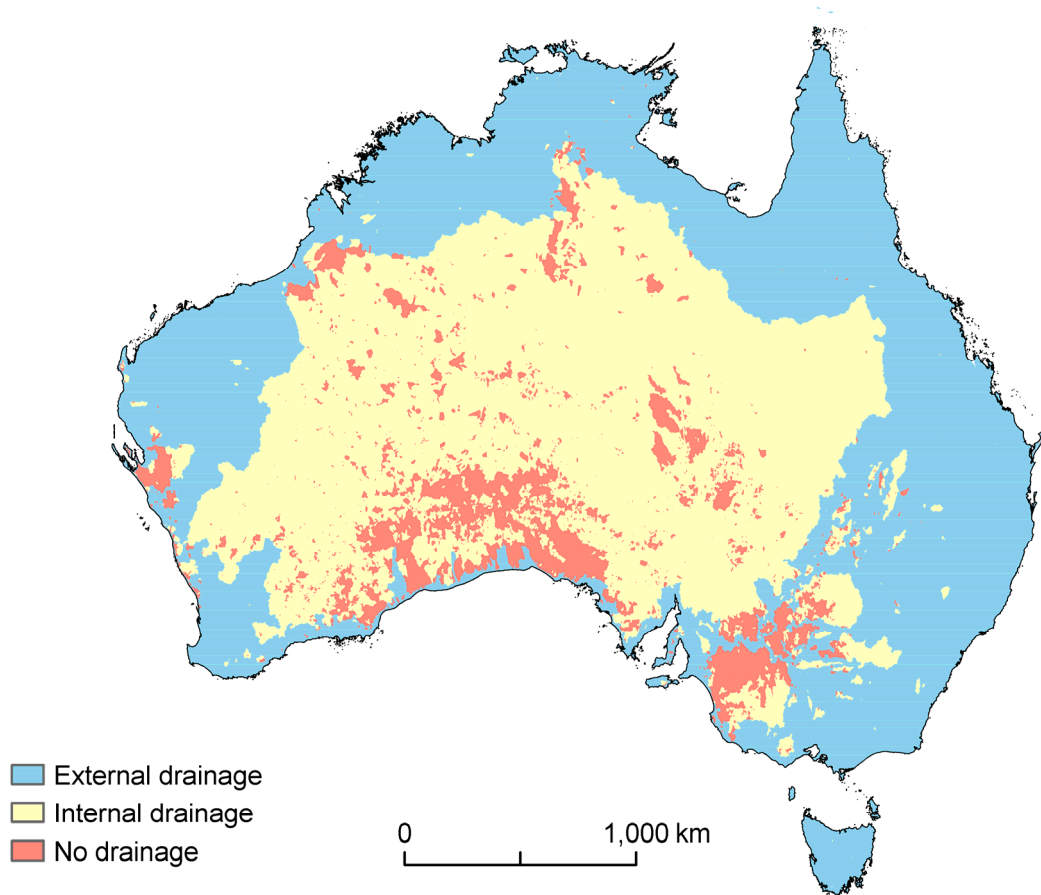


Figure 5.7 Drainage classes. Drainage basins with no DEM-derived streams were mapped as “No drainage”.

5.2. Results

5.2.1. Drainage basins

Nearly 50,000 drainage basins were identified. Each was directly associated either with an outlet on the coast (46% by area) or an inland sink, for example, a terminal lake, clay pan or depression in a karst landscape (54% by area) (Figure 5.7). However, the majority (67%) of these basins are very small, less than 10km² in area. These small basins usually occur as the areas draining to clusters of clay pans or small dry lakes or on the coastal fringe, as the grid cells that drain directly to the sea rather than through a river mouth. Few of these small basins have a well-developed stream network – the 9-second DEM-derived network is represented in just 8% of these small basins. In contrast, most of the large basins are drained by major river systems, although several are associated with a terminal lake (Table 5.1). Another 704 basins with an area greater than 1,000km² drain the catchments of smaller rivers and inland lakes.

Table 5.1 Australia's largest drainage basins (area greater than 25,000km²)

Name	Area (km ²)	Name	Area (km ²)
Lake Eyre	775,674	De Grey River	54,834
Murray-Darling	763,589	Daly River	49,931
Norman/ Flinders/ Gilbert Rivers	231,324	Lake Gregory	48,862
Fitzroy River (Qld)	141,341	Hanson River	48,681
Burdekin River	129,709	Fortescue River	48,589
Gascoyne River	102,719	Lake Mackay	44,632
Fitzroy River (WA)	90,467	Warne River/ Yarra Yarra Lakes	44,163
Victoria River	89,525	Bulloo River	41,620
Ord River	83,801	Lake Frome	40,635
Edward / Mitchell River (Qld)	81,813	Lake Disappointment	34,518
Leichhardt River	79,751	Burnett River	33,299
Ashburton River	75,029	Hay River	32,658
Swan River	73,569	Lake Torrens	31,427
Roper River	65,521	Lake Woods	28,990
Murchison River	65,384	Playford River	25,656

Much of the broader scale drainage structure of the continent is only revealed after the internally draining basins are iteratively merged with a lower neighbouring basin via their lowest saddle-point (Figure 5.8). The majority of basins have a saddle point that is less than 10m above the sink but a small number were only associated with another basin by application of a 100m depth threshold (Table 5.2). In this way, all internally draining, closed basins were associated with a portion of the coast or the Lake Eyre drainage basin.

These results were compared to two other assessments:

- the widely adopted AWRC river basins and drainage divisions (Australian Water Resources Council, 1976) and their most recent revisions (AUSLIG, 1997; Geoscience Australia, 2003a) (Figure 3.1) and
- the drainage analysis of Hutchinson and Dowling (1991) based on a DEM with a grid resolution of 0.025 degree of latitude and longitude (approximately 2.5km).

Table 5.2 Number of basin units remaining after aggregation

Sink depth threshold (m)	Number of basin units	Basin unit area (km ²)	
		Mean	Maximum
No aggregation	49,688	155	775,674
10	31,503	245	853,832
20	29,692	260	1,070,561
30	29,338	263	1,217,825
40	29,268	263	1,222,414
50	29,254	264	1,222,414
100	29,244	264	1,310,508

The latter analysis was not available in a digital form. Still, no major departures from the current analysis could be detected and there is broad agreement on the principal drainage divisions. There are, however, some significant departures from the AWRC boundaries, most notably for the Western Plateau Drainage Division and the rivers flowing into the Gulf of Carpentaria (Figure 5.8). The drainage basins that are grouped by the AWRC Western Plateau Drainage Division are associated with three different sections of the coast. Consistent with the analysis of Hutchinson and Dowling (1991), the Western Plateau Division is split into a northern and southern section by a major drainage divide. The AWRC basins Mackay, Wiso and Barkly, included within the Western Plateau Division, are linked by this analysis to coastal basins in the AWRC Timor Sea Drainage Division.

The coding of flow bifurcations delineates connected drainage areas that were identified as separate basins by AWRC. For example, the Norman, Gilbert and Flinders Rivers, draining to the Gulf of Carpentaria, delineate a single drainage basin in this analysis but AWRC divides them into three separate basins with boundaries drawn through floodplain distributary channels (Figure 5.8). The new drainage analysis recognises substantial areas of internal drainage within the Murray-Darling Basin. As a result, it finds the total area draining to the mouth of the Murray River to be nearly 300,000 km² less than the usually quoted area of 1,061,469 km² (http://www.mdbc.gov.au/about/basin_tour).

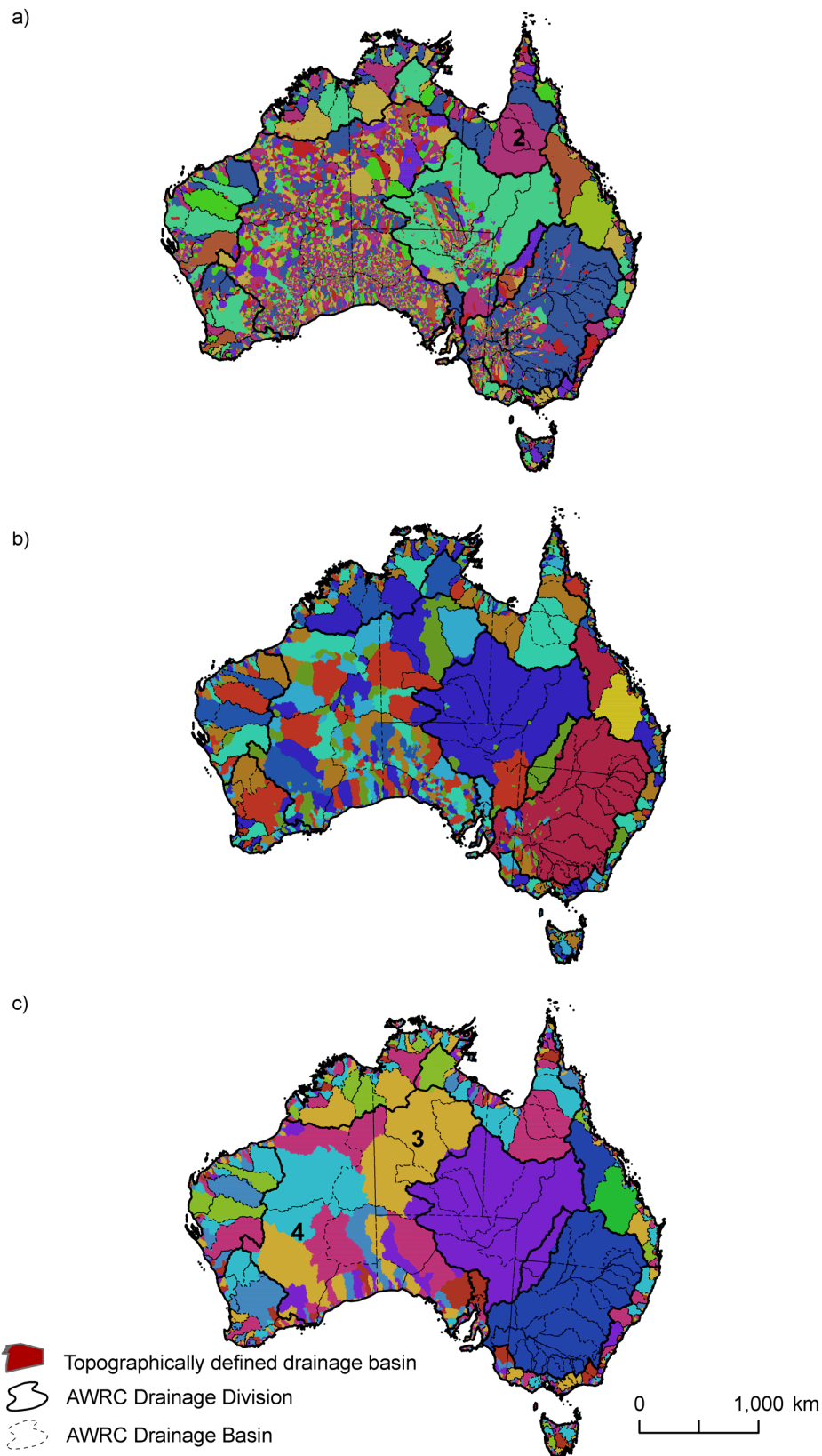


Figure 5.8. a) The topographically defined drainage basins of Australia delineated by drainage analysis of the 9 second DEM; b) Internally draining basins combined with a neighbouring catchment via their lowest pour-point (where sink depth < 20m); c) Internally draining basins combined to a sink depth of 100m. Notable changes from AWRC: 1) areas of internal drainage within the Murray-Darling Basin; 2) Basins of rivers connected at high flows through a series of distributaries are combined, for example, the Norman, Gilbert and Flinders Rivers; 3) association of the AWRC basins Mackay, Wiso and Barkly with the Timor Sea Division; 4) subdivision of the AWRC Western Plateau Drainage Division into northern and southern sections

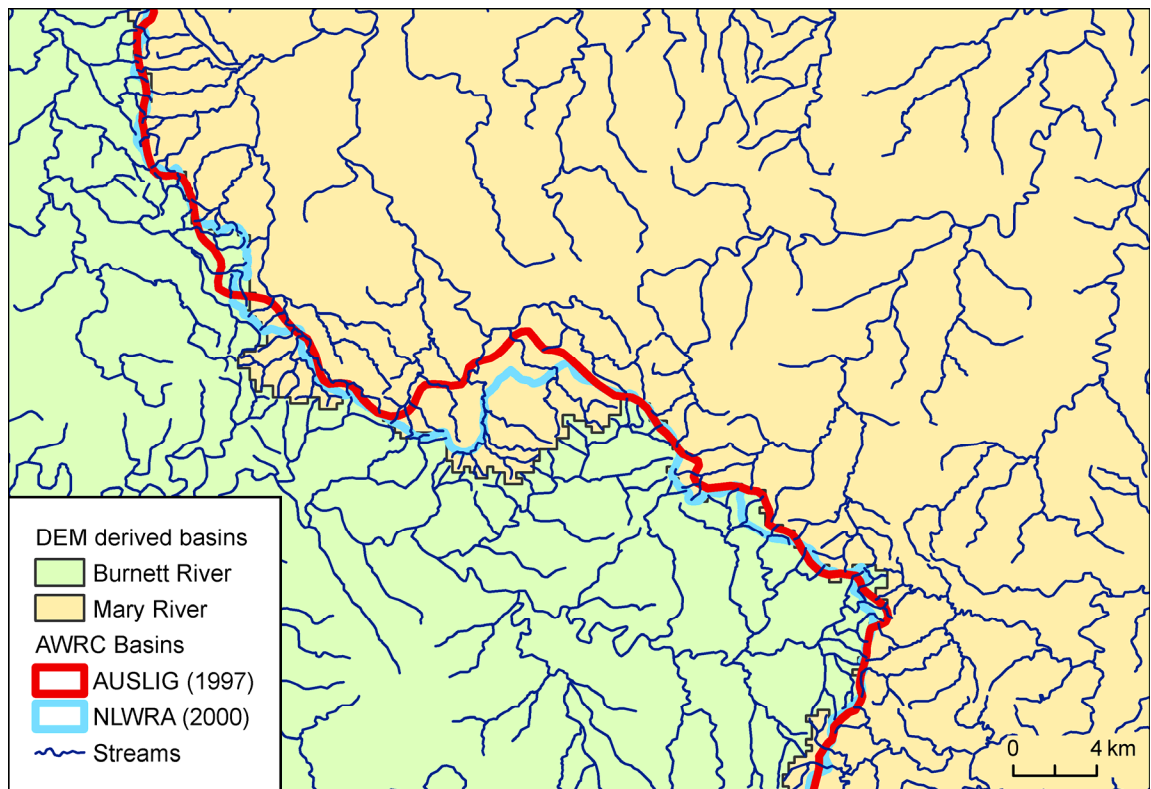


Figure 5.9 Improved placement of drainage divides: basin boundaries delineated by this study for the Burnett and Mary River (Queensland) with recent versions of the AWRC basins; Australia's River Basins (AUSLIG, 1997) and Surface Water Management Areas compiled for the NLWRA (Geoscience Australia, 2003a).

Other departures from the AWRC boundaries can be attributed to improved placement of drainage divides or the use of different catchment outlets to seed drainage basins. This analysis corrects serious errors in the placement of drainage divides by AWRC (Figure 5.9), though boundaries are more generalised than those of AWRC that were delineated from larger scale mapping (at scales as fine as 1:10,000, AUSLIG, 1997). As described previously (Section 3.1) AWRC boundaries sometimes delineate convenient administrative units rather than true topographically defined drainage basins and hence are not comparable with the topographically defined basins derived by this analysis. However, even where topographically defined, the AWRC basins may be substantially different, depending upon the placement of the basin seeds. For instance, this analysis places a basin seed to the north of Adolphus Island in the Cambridge Gulf in north-west Australia (Figure 5.10). The delineated basin incorporates the catchment areas of both the Pentecost and the Ord Rivers. In contrast, the AWRC basins divide the Ord River basin into the catchments of the Ord and Pentecost Rivers, effectively seeding a basin at the confluence of the Pentecost River and the west arm of the Ord River.

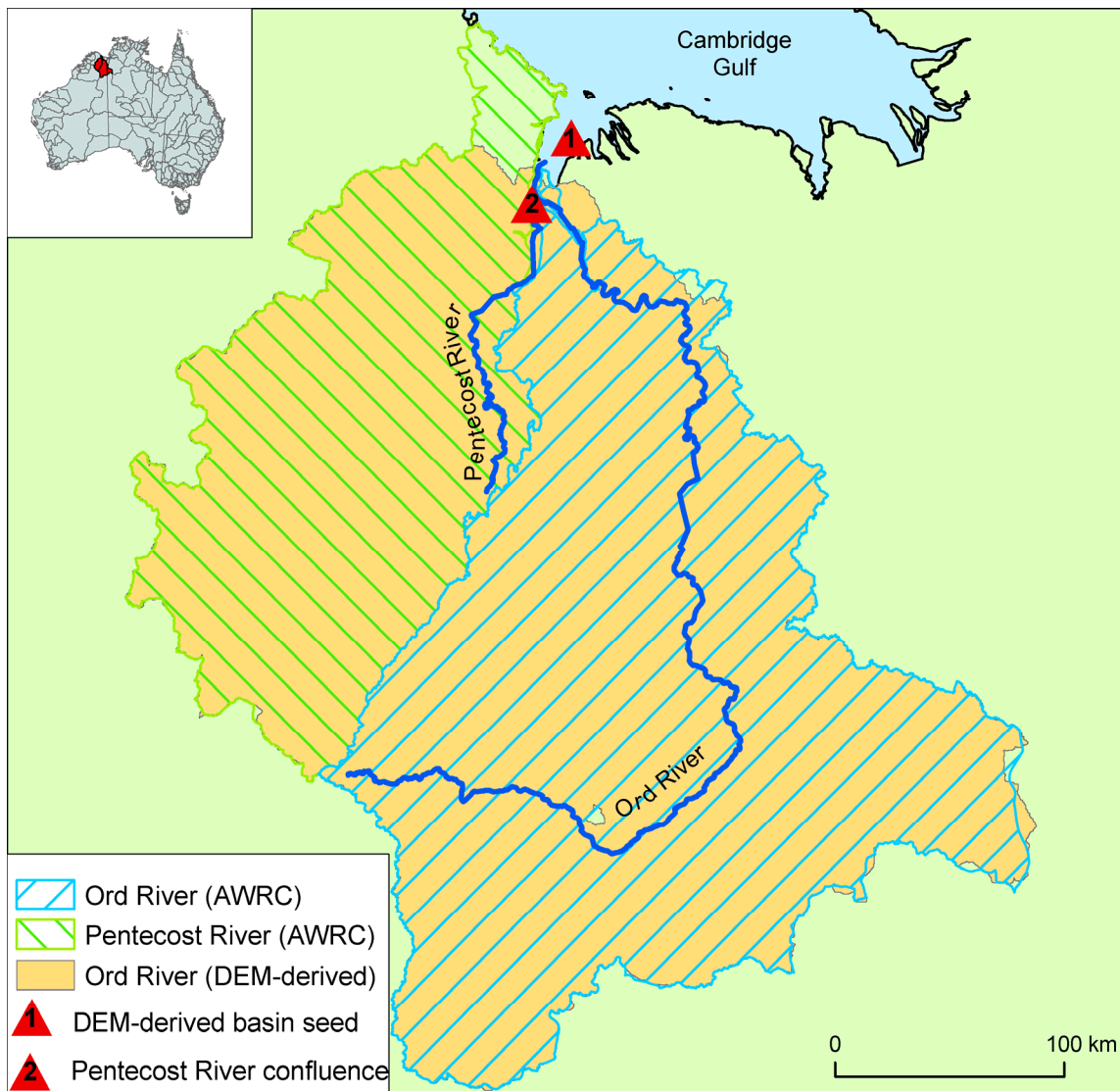


Figure 5.10 Comparison of drainage basins derived in this study and AWRC: effect of catchment outlet choice. The coastal outlets identified from the DEM flow pathways seed the Ord River Basin shown in orange. In contrast, AWRC sub-divides the Ord River basin into the Pentecost and Ord Rivers, effectively seeding a basin at the confluence of the Pentecost and Ord Rivers. Note also that the AWRC basin Pentecost River also includes an adjacent coastal area that is outside of the topographically defined catchment.

5.2.2. Nested catchments

A 9 level Pfafstetter sub-division of the continent was achievable at 9 second resolution (Table 5.3). The average area of the catchment units changes little after Level 6 as many basins were already sub-divided as far as possible. While the numbers and average area of catchment units at each level are very similar whichever method was used to differentiate tributary and mainstems (Table 5.3), the pattern of basin sub-division may be very different (Figure 5.11). The modelled water balance produces results more consistent with usual conventions, for example, treating the Darling River as a tributary of the Murray River even though it drains the larger area.

Table 5.3 Properties of the 9 level Pfafstetter sub-divisions.

Tributary / mainstem discrimination				
Level	Contributing area		Catchment water balance ("flow")	
	No. units	Average area (km ²)	No. units	Average area (km ²)
1	10	771,179	10	771,179
2	100	77,105	100	77,105
3	878	8,782	859	8,976
4	6,736	1,144	6,500	1,186
5	40,167	192	39,098	197
6	165,393	47	158,525	49
7	424,821	18	400,635	19
8	685,777	11	660,097	12
9	817,717	9	803,037	10

At any level in the Pfafstetter hierarchy the size of the catchment unit varies greatly. However, it is relatively straightforward to extract units of a more consistent size for any particular application by identifying the basin specific level of sub-division that produces units with an average area closest to the desired size. For instance, to extract catchment units of about 500km² in area the user would first calculate for each basin the average area of units at each level in the nested catchment hierarchy. The level that has an area closest to 500km² is flagged and the catchment units extracted from this basin-specific level of sub-division (Figure 5.12). The required level of sub-division necessarily varies between drainage basins, depending upon the size of the drainage basin and the level at which it is first independently coded as a basin (i.e the first level for which it has an even digit or zero).

The extent of drainage basin sub-division varies greatly at any level in the hierarchy. The larger basins are sub-divided at high levels in the nested hierarchy while small basins are still aggregated into inter-basin units. Again, if an application requires a particular level of sub-division, the appropriate basin-specific level to use can be determined directly from the Pfafstetter codes. For example, a nested 3 level split is generated by extracting the catchment

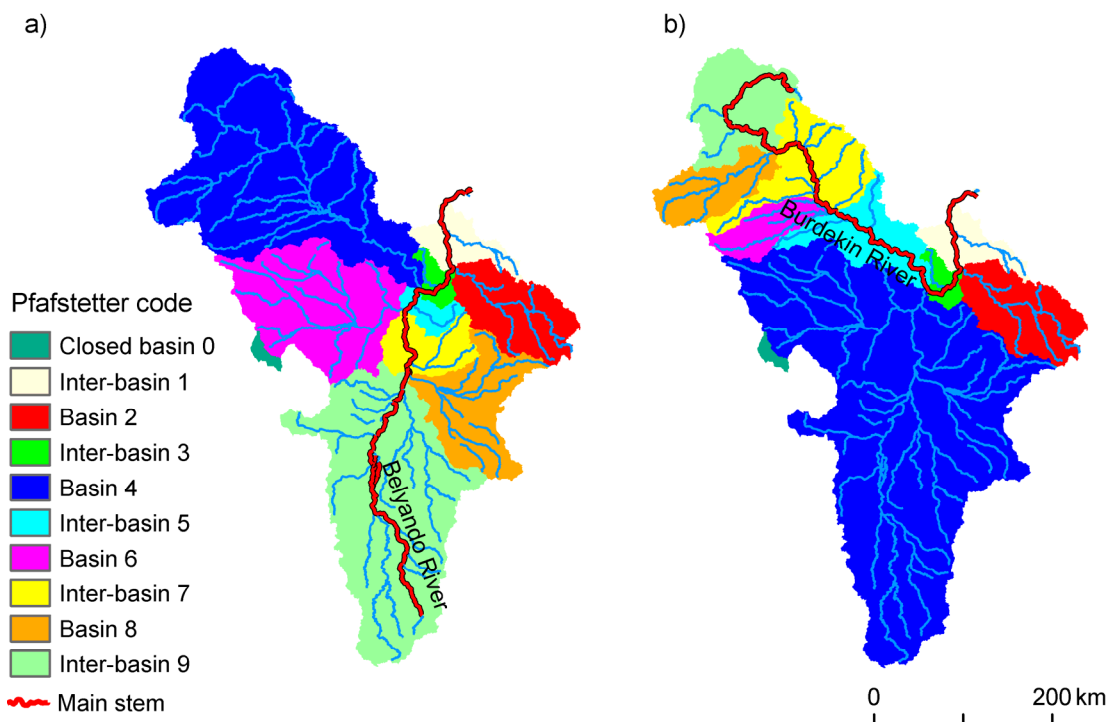


Figure 5.11 Level 2 Pfafstetter sub-division of the Burdekin River basin, Queensland: tributary-main stems differentiated according to a) contributing area and b) catchment water balance. Consistent with usual convention the catchment water balance identifies the Burdekin River as the main stem

units at the Pfafstetter level $n+3$ where n is the first level at which the drainage basin is separately coded.

The most useful property of the Pfafstetter coding is its ability to indicate topological relationships in a catchment. Network position is inferred from the ordinal value of a digit and by whether it is odd or even. A larger digit indicates a section of river further upstream while an even digit designates a tributary off the main channel. With simple algebraic queries on the Pfafstetter codes it is possible to determine whether or not a river section is likely to be affected by a particular activity without reference to a map or flow routing functions in a GIS. Figure 5.13 illustrates a hypothetical example for a Level 4 Pfafstetter basin 4284. A new dam is proposed for the tributary stream in catchment number 42846. The streams that would be affected by the dam (all main stem channels downstream of the dam) can be identified by querying the Pfafstetter codes. First, they must match on all leading even digits. Trailing odd digits must also be lower. For example, the stream flowing through inter-basin 42841 would be affected because the match on the leading even digits (4284) indicates they are within the same basin while the trailing digit (1) is less than that of the tributary stream on which the dam is to be located (6). Although also matching on the leading even digits, streamflows in 42849 are not impacted by the dam because the trailing digit (9) is greater than 6, implying that the section of stream is upstream of the dam site. Likewise, the stream in catchment unit 42844 is also unaffected because the trailing digit, though lower, is even signifying it is a tributary off the main stem.

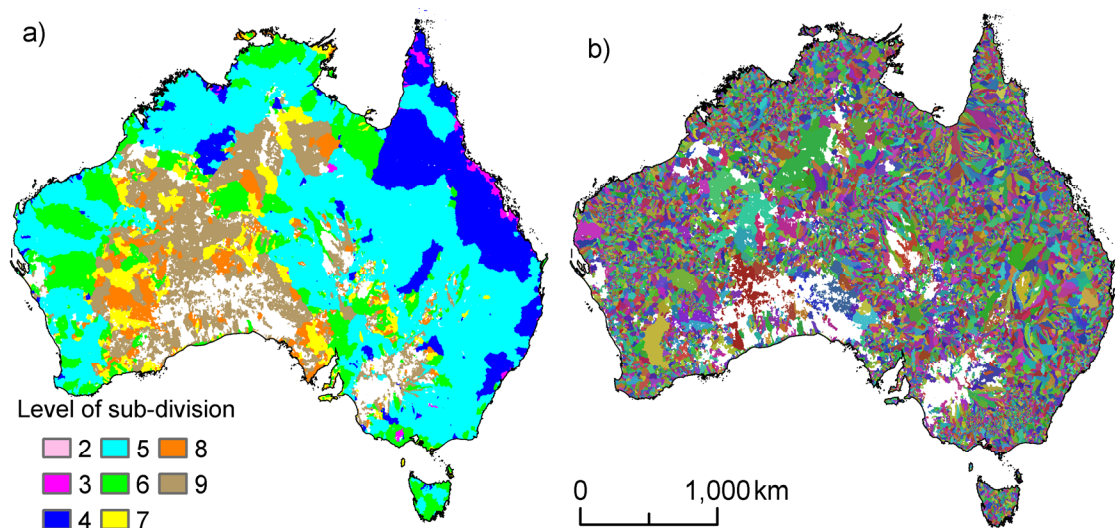


Figure 5.12 Extracting catchment units of a consistent area from the Pfafstetter nested catchment hierarchy: a) the level of sub-division at which units have a basin average area of 500km² and b) hydrologic units extracted at this basin specific level. Entire basins were extracted for basins not sub-divided by Level 9. No drainage areas are shown in white.

5.3. Discussion

5.3.1. A new nested catchment reference system

Application of the new drainage analysis procedures in association with the Pfafstetter scheme produced a new stream and catchment reference system and a spatial framework suitable for conservation planning tasks at a range of spatial scales. For example, drainage basins are the logical unit for the development of biogeographic regionalizations (Unmack, 2001; Tait *et al.*, 2003) while the larger catchments might be appropriate units for reporting progress towards conservation goals and for application of management prescriptions. Smaller sub-catchments could be a sensible choice as planning units for application of systematic conservation planning algorithms. Greater differentiation of intra-catchment conservation values is achievable by individually characterising and classifying stream segments, the smallest units in the spatial framework.

The multi-scale framework is thus flexible in its application, enabling the scale of analysis to be adjusted to that most appropriate for the patterns of interest and/or the available data. It supports broad scale national planning while setting the context for local, regional or catchment scale assessments. It also provides a mechanism for data sharing and communication across jurisdictional borders.

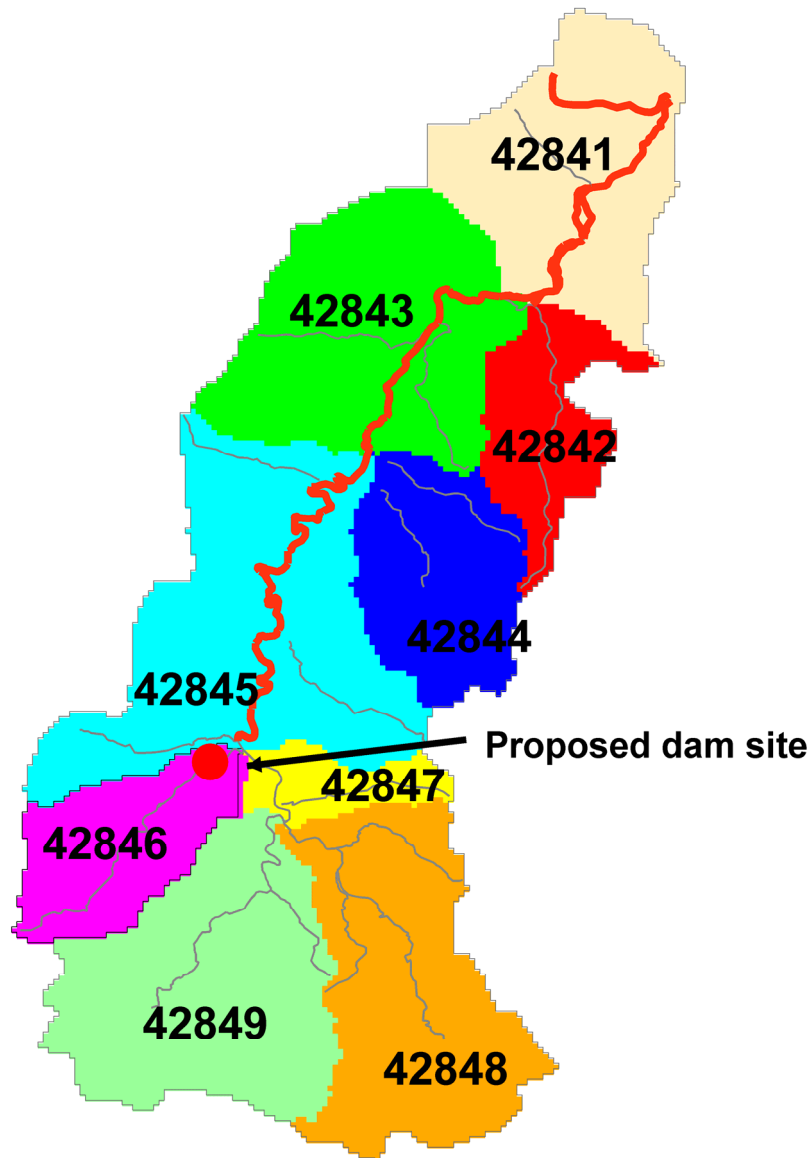


Figure 5.13 Using the Pfafstetter codes to infer the influence of a hypothetical new dam. The stream segments that would be affected by the proposed dam are shown in red.

Due to the recognition of anabranches that connect rivers at high flows the placement of some drainage divides depart significantly from those previously derived (Hutchinson and Dowling, 1991; AUSLIG, 1997; Hutchinson, Stein *et al.*, 2000; Geoscience Australia, 2003a). By removing artificial divides between such drainage systems the new drainage basin delineations will provide an improved basis for the development of biogeographic regionalizations.

The basin boundaries of coastal-draining systems are generally in close agreement with the topographically defined boundaries in the widely adopted analysis of the Australian Water Resources Council (1976) and subsequent updates (AUSLIG, 1997; Geoscience Australia, 2003a). The more significant departures are largely attributed to more accurate placement of drainage divides in the current study. Smaller differences arise due to the limitations of the grid

data structure and resolution employed in this analysis. The AWRC basins were often mapped at finer scales (ranging from 1:10,000 to 1:250,000) (AUSLIG, 1997; Geoscience Australia, 2003a) and thus are less generalised.

In contrast, this drainage analysis suggests some significant changes to the AWRC drainage divisions and basins in inland areas, although these are yet to be systematically documented and reviewed. Validation of boundary changes is limited by the availability of suitable, independent reference data. The 1:100,000 scale topographic map series is the finest scale available for many areas but at this scale drainage divides are poorly defined in areas of low relief. While a continent-wide reference set was available for corroboration of the derived stream network, the reference sets used for validation of catchment boundaries were biased towards higher relief, upper catchment positions, where drainage divides and surface flow paths are expected to be well defined. This is likely to be of little consequence for continental-scale river conservation planning, however, as rivers are largely absent from the arid drainage basins with the most significant boundary changes. Nevertheless, a known source of potential error is the omission of sand ridges from the 9-second DEM representation of terrain features (Hutchinson *et al.*, 2001). Thus catchment delineations should be used with caution in areas of extensive dunefields.

The division of the Western Plateau Drainage Division into northern and southern sections, together with the incorporation of the AWRC basins Mackay, Wiso and Barkly with the Timor Sea Drainage Division, corroborates the earlier drainage analysis of a 0.025 degree resolution DEM by Hutchinson and Dowling (1991). These authors found that the morphology of this drainage division could be related to descriptions of formerly external, palaeodrainage systems. The removal of most spurious sinks during DEM development averted the need for automatic sink filling procedures and consequent loss of information on terrain shape. Conversely, by retaining natural depressions in the DEM, large numbers of endorheic drainage basins were identified. This contrasts with the drainage patterns of other continents. Endorheic basins, areas currently isolated by surface flows and therefore potentially containing unique biodiversity, occupy more than half of the Australian continent but just 5% of the North American continent or approximately 27% of the global land area (Battle Creek Watershed Conservancy, <http://www.battle-creek.net/glossarye.html>, accessed 5/8/05).

5.3.2. The Pfafstetter system

The nested catchments derived according to the Pfafstetter system offer a number of enhancements over previous continental scale catchment frameworks. The topological information conveyed by the Pfafstetter codes (up/downstream, tributary/main stem, higher level catchment membership) provides the ability to rapidly identify conservation values potentially affected by human activities or management actions without need for sophisticated

GIS analyses. The hierarchical nature of the system, reflected in the hydrologic unit codes, facilitates application across a range of spatial scales. The coding system is also self-replicating so that where higher resolution DEMs are available, finer level sub-divisions can be incorporated without alteration of existing codes.

However, because the Pfafstetter system limits the number of divisions at each level it produces a great disparity in the size of hydrologic units, particularly at higher levels in the hierarchy. The lack of a consistent relationship between the approximate size of the units and the hierarchical level is perceived as a major disadvantage (Britton, 2002) but one that was shown (Section 5.2.2) could be overcome by selecting a drainage basin specific hierarchical level. Applying an independent first level division, similar for example to the AWRC drainage divisions, would reduce the large disparity in the size of inter-basin units at the highest levels with the added advantage of facilitating linkages with existing databases built upon the AWRC framework.

The extensions to the Pfafstetter system introduced in this study provide for greater flexibility and increase the likelihood that the catchment framework will be accepted by conservation planners. Allowing basin sub-division as long as there was at least one tributary enabled a finer scale sub-division of some basins than would have been the case had the original scheme's requirement for four tributaries been strictly enforced. Additionally, using a surrogate for flow, rather than catchment area, to discriminate tributaries and main-stems, delineates hydrological units that are more consistent with usual conventions and therefore, more likely to be adopted.

However, the concept of a main-stem and tributary is not well defined in extensively anabranching river systems and alternative methods may need to be devised to provide a sensible sub-division of these systems. Uncoordinated drainage patterns are also not well catered for by the Pfafstetter scheme. The connected stream network derived by tracing surface flow paths effectively "joined up" the isolated elements of the network and provided a reasonable solution, but the additional confluences that resulted occasionally disrupted the hierarchical nesting of hydrologic units. Thus, where stream segments were broken by a confluence with a "joined-up" tributary, the segment could be a member of more than one higher-level catchment unit.

Under the Pfafstetter scheme adjacent smaller coastal drainage basins are grouped as inter-basins. Drainage areas on islands are assigned to the nearest mainland inter-basin. As a result smaller drainage basins in Tasmania were still grouped with mainland drainage basins as low in the hierarchy as level 4. In contrast, the AWRC framework separates Tasmanian drainage basins from those on the mainland at its highest level, the Drainage Division. This need not be a problem for most analytical uses as drainage basins can be individually extracted from lower levels in the hierarchy, but it may be a constraint on acceptance of the new catchment reference system for reporting and management applications currently employing the AWRC framework

and another argument for adopting modified AWRC drainage divisions as the first level sub-division of the continent.

Inclusion of the very large number of small, endorheic drainage basins was also problematic under the Pfafstetter scheme. At each level only the largest closed basin in a hydrologic unit was identified; remaining closed basins were associated with a lower neighbouring catchment unit. As a result, many closed basins were not sub-divided even at level 9 though many such basins did not have a well-developed drainage network as a basis for sub-division. If required, closed basins could be flagged and excluded from a catchment-unit for analysis. Such tasks will require users be provided with comprehensive documentation. Ideally they would also have access to procedures for querying the catchment framework and extracting units meeting desired specifications such as size, spatial extent and level of sub-division. This might be done through an online mapping tool or dedicated software distributed with the data.

5.4. Conclusions

A new stream and catchment reference system was developed to serve as a spatial framework for systematic conservation planning for Australian rivers and streams. The new framework overcomes many of the shortcomings of existing continent-wide catchment delineations that make them unsuitable for conservation planning applications. It respects the true drainage structure of the Australian continent as portrayed on topographic maps and addresses the conservation planning requirements of rivers and streams. Hierarchical relationships among the consistently defined spatial units are explicitly represented, facilitating application at a range of spatial scales and recognition of linkages between elements of the fluvial system. The catchment framework can also be linked to higher resolution state or regional data for finer scale (e.g. catchment) planning and management.

Chapter 6. River classification: a review

Classification is the process of identifying groups of objects or “types” that are relatively homogenous for one or more characteristics. If the objects are spatially referenced or the classification criteria can be applied to them the groups may be mapped, but the groups need not be spatially contiguous. Regionalization, on the other hand, is a particular form of spatial classification, where boundaries are drawn around relatively homogeneous geographic areas (Bryce and Clarke, 1996).

Classification is important for conservation planning. It provides the necessary context to:

1. evaluate conservation value criteria (e.g. rarity, naturalness, diversity, representativeness) (Austin and Margules, 1986; Boon, 1992);
2. assist description of a reference state for assessment of ecological condition (Schmutz *et al.*, 2000; Whittington *et al.*, 2001);
3. characterize regional patterns of ecosystems (Cleland *et al.*, 1997);
4. define conservation targets for inclusion in a comprehensive system of protected areas (Sowa *et al.*, 2004; Higgins *et al.*, 2005);
5. capture and organize complex information efficiently (Uhlig and Jordan, 1996);
6. extrapolate site specific studies (Hurley and Jensen, 2001);
7. enhance sharing and communication of information (Cleland *et al.*, 1997; Sowa *et al.*, 2004);
8. facilitate cooperation and coordination amongst jurisdictions (Pigram and Sundell, 1997); and
9. define ecologically meaningful planning regions (Yencken and Wilkinson, 2001; Abell *et al.*, 2002).

While there are regional (Riley *et al.*, 1984; Fatchen and Lustig, 1986; Macmillan and Kunert, 1990; Olsen and Skitmore, 1991; Whittington *et al.*, 2001) or statewide (Hughes and James, 1989; Land Conservation Council, 1989; Doeg, 2001; Jerie *et al.*, 2003; Kingsford *et al.*, 2004) classifications of Australia rivers and floodplain wetlands, there is no nationally consistent mapping of stream types (Schofield *et al.*, 2000). Accordingly, this chapter explores the prospects for a continent-wide classification of Australian rivers and streams for conservation planning application.

The chapter begins (Section 6.1) with a brief overview of the conceptual underpinnings of common approaches to classification. This is followed by a review of river classification

systems, divided into three broad categories reflecting the attributes used to discriminate groups: biological (Section 6.2); habitat (Section 6.3) and landscape classification (Section 6.4). While there have been many reviews of river classification (Mosley, 1987; Gordon *et al.*, 1992; Naiman *et al.*, 1992; Downs, 1995; Thorne, 1997; O'Keefe and Uys, 2000; Jensen, Goodman, Frissell *et al.*, 2001; Makaske, 2001), few have focussed on large-scale conservation planning applications or applicability to Australian drainage systems. Hence, this review will concentrate on the classification needs of continental-scale conservation planning and, wherever possible, illustrate the approach with Australian examples. The chapter concludes with recommendations for development of a classification for conservation planning for Australian rivers and streams.

6.1. Classification approaches

Classifications may be applicable at a single scale or link a range of scales within a hierarchical framework. Stream order (Strahler, 1957) is a simple but enduring single-scale classification scheme that measures relative position in the drainage network. It may be a useful indicator of stream segment biotic and physical characteristics within a drainage basin, but it is much less reliable at regional scales (Naiman *et al.*, 1992). Similarly, longitudinal zonations of rivers using the presence of one or more species (examples in Mosley, 1987) rarely apply beyond the regions in which they were developed (Naiman *et al.*, 1992). Hierarchical models of stream classification place streams explicitly in the context of their linkages with the surrounding drainage catchment and terrestrial landscape (Jensen, Goodman, Frissell *et al.*, 2001) and as a result are effective across a range of spatial scales from local to continental.

The hierarchical structure in ecological classification may be developed by agglomeration or by sub-division (Bourgeron *et al.*, 2001a). The divisive “top down” approach starts from large, ecologically heterogeneous areas that are successively divided into the more homogeneous lower levels of the ecological hierarchy. Top down approaches have been widely adopted for hierarchical river classification (e.g. Frissell *et al.*, 1986; Blackman *et al.*, 1992; Maxwell *et al.*, 1995; Calvert *et al.*, 2001; Snelder and Biggs, 2002; Higgins *et al.*, 2005). In a top down approach, ecological units are mapped at progressively increasing resolution and finer map scale with associated increases in data and analysis requirements. The top down approach is appealing because classifications can be produced for entire continents with limited data, providing the context for finer scale delineations in selected areas (Cleland *et al.*, 1997). It also accords with theories of the hierarchical organization of ecological systems (O'Neill *et al.*, 1989) and the asymmetry in the relationship between hierarchical levels. Thus, while higher levels of the system functionally constrain the lower levels, emergent properties of higher levels cannot be

predicted from the properties of lower levels (Zonneveld, 1994; Perera *et al.*, 1996; Bourgeron *et al.*, 2001a).

In practice however, a top down approach has several limitations. The emergent properties of drainage basins, the highest levels in the natural hierarchical organization of river ecosystems (Section 2.3), may be difficult to characterize for classification purposes. Differentiating criteria are applied using summary values, but basin average values may be meaningless for large and heterogeneous basins. Climatic criteria are often used to define the classes of higher levels in hierarchical ecosystem classifications (Bailey, 1996; Cleland *et al.*, 1997; Calvert *et al.*, 2001; Snelder and Biggs, 2002). However, these are not readily applied to large river basins that may include climatic types from montane to tropical lowland (Omernik and Bailey, 1997). Similarly, a description of flow patterns at the catchment outlet has little relevance for biota living in the headwater streams. A top down approach starting with a grouping of river basins may obscure similarities more closely associated with particular habitat characteristics than patterns of drainage system connectivity, for example, distinctive montane fish communities that span drainage divides (Gehrke and Harris, 2000). Sequential divisive approaches are also sensitive to the order in which classification criteria are specified (Phillips *et al.*, 2002) which may vary in importance with, for example, the nature of the valley substrate (Cohen *et al.*, 1998; Heritage *et al.*, 2001).

In contrast, agglomerative or “bottom up” approaches progressively integrate the objects of the classification according to their shared similarities from the lowest levels in the hierarchy to increasingly higher level classes. The bottom up approach is consistent with a view that the character and behaviour of the river reflects the collective characteristics of the tributary sub-catchments (Brierley and Fryirs, 2002). It is inductive and independent from spatial constraints (Bourgeron *et al.*, 2001a) except through the spatial dependence of the classificatory variables. However, bottom up approaches depend on data being available at the finest scales for a stream to be classified at any level in the hierarchy.

The attributes used to discriminate groups may be direct measures of river ecological characteristics (“response” variables) or the principal factors responsible for these characteristics (“driving” or “controlling” variables) (O’Keefe and Uys, 2000; Bourgeron *et al.*, 2001b). Response variables include descriptors of biotic community structure (taxon distribution and abundance) or habitat descriptors, principally hydrological or channel geomorphic characteristics. Naiman *et al.* (1992) recognise controlling factors operating across a range of spatial and temporal scales. Ultimate controls, regional geology, climate and zoogeography, operate across large areas and can be stable over long time scales (centuries to millennia). Proximate controls include the local geomorphic processes (e.g. channel migration, sediment transport) and biotic processes (e.g. reproduction, competition, predation) that alter

river characteristics over short time periods (decades or less) and are important at small spatial scales.

6.2. Biological classification

Classifications based on the distribution patterns of the biota are often favoured for conservation applications (Olson *et al.*, 2001; Tait *et al.*, 2003). Dominant species, indicator species or assemblages of organisms (commonly fish or macroinvertebrates) may be used to differentiate groups (Gordon *et al.*, 1992). For example, Gehrke and Harris (2000) identified four broad groups of rivers, Montane, North Coast, South Coast and Murray-Darling, from the fish assemblages recorded in a systematic survey across NSW. Similarly, Marchant *et al.* (1999) classified Victorian rivers according to the macroinvertebrate communities found in edge and riffle habitats at 199 sites.+

However, while systematic conservation planning requires that a classification be mapped for all rivers and streams, biological survey data are inevitably restricted to a small number of sites. Wider spatial distributions may be predicted from the site records by linking these data to mapped environmental variables using bioclimatic modelling techniques such as BIOCLIM (Nix, 1986) (e.g. Fischer *et al.*, 2001; Lindenmayer *et al.*, 2002) or a range of statistical modelling methods (Austin, 1985; 2002; Thuiller, 2003). The modelled distributions of terrestrial species and communities have been used in conservation planning exercises at both regional (Margules and Nicholls, 1987; Margules and Stein, 1989; Ferrier, Drielsma *et al.*, 2002; Ferrier, Watson *et al.*, 2002; Scott and Drielsma, 2003) and national (Nix *et al.*, 2000; Faith, Margules, Walker *et al.*, 2001) scales. Similar techniques may also be employed to model fish occurrence (Joy and Death, 2002; Olden and Jackson, 2002). Pusey *et al.* (2000) for example, demonstrated that the structure of fish communities in four Queensland streams could be predicted from regional or catchment scale factors.

If suitably representative site data with matching environmental attributes exist (Mackey *et al.*, 2001) models could be developed for other riverine taxa, but achieving the level of sampling in both space and time required to model the continental distribution of highly dynamic taxa is problematic. For instance, variation in macroinvertebrate community structure occurs at fine spatial (Robson and Chester, 1999; Dovciak and Perry, 2002; Boyero, 2003) and temporal (Scarsbrook, 2002) scales in relation to intra- and inter-annual discharge variability (Humphrey, 1997) or local disturbance (Matthaei and Townsend, 2000). Survey sites are seldom 'representative' of larger spatial units, such as stream segments (Downes *et al.*, 2000). The AusRivAS bioassessment program (<<http://ausrivas.canberra.edu.au/index.html>>) developed statistical models that could predict the expected macroinvertebrate community at a test site, but the models require both coarse-scale, map-based variables and local scale, field measures of dynamic properties including water quality and substrate composition (Smith *et al.*, 1999; Turak

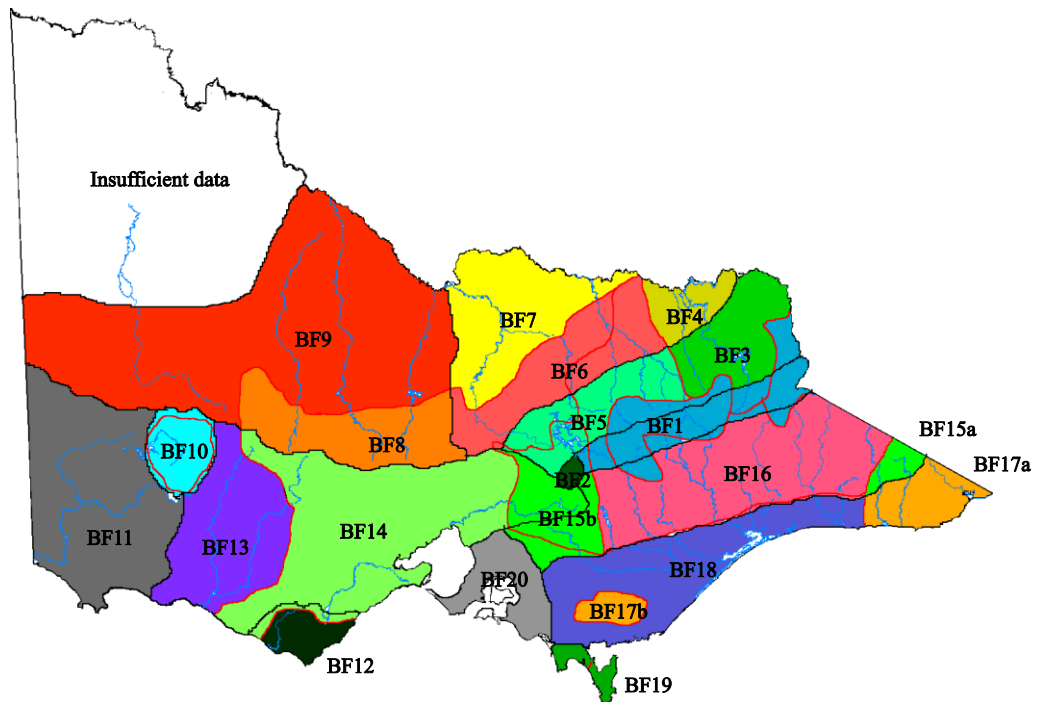


Figure 6.1 Riverine Biological Regions of Victoria (after Doeg, 2001)

et al., 1999; Read, 2001). Models of macroinvertebrate community structure developed using remotely mapped environmental variables alone are less successful - less than half of the regional models for the NLWRA Assessment of River Condition were considered satisfactory (Norris *et al.*, 2001).

In lieu of comprehensive data or modelled distributions, bioregionalisations are derived using expert judgement to place boundaries around broad regions of relatively homogeneous biological assemblages. For example, the boundaries that separate Victoria's macroinvertebrate river regions were positioned by professional judgement (Wells *et al.*, 2002), guided by environmental features and macroinvertebrate community site groups, following the method of Newall and Wells (2000). These regions were combined with another regionalisation developed from fish distribution maps, also using expert judgement (Doeg, 2001). Unfortunately, the resulting Riverine Biological Regions (Figure 6.1), that underpin the selection of representative rivers in Victoria, required "a considerable amount of latitude" to reconcile the combined boundaries (Doeg, 2001) and with limited data, statewide coverage was not possible.

Regionalisation assumes that community similarity most strongly reflects geographic proximity but other trends, such as stream longitudinal gradients, may be superimposed (Marchant *et al.*, 1999) to produce complex distribution patterns only weakly reflecting geographic proximity (Turak *et al.*, 1999; Heino *et al.*, 2003). The degree of spatial organization in community structure that is detected is both a function of the spatial scale over which data are gathered (Marchant *et al.*, 1999) and the accepted level of within group heterogeneity.

Biogeographic methods offer an alternative approach to bioregionalisation with region boundaries emerging from concordant cross-taxa disjunctions in distribution patterns, rather

than community similarity. Boundaries delineate areas having a distinctive recent evolutionary history (Unmack, 2001) and reflect the influence of ultimate controls (Naiman *et al.*, 1992) (e.g. major geological events, glaciations, and land bridges) on aquatic ecosystems, constraints that may be difficult to incorporate into species' or communities' distribution models using abiotic factors alone. Biogeographic regionalisations commonly define the highest levels in a hierarchical classification scheme (Naiman *et al.*, 1992; Maxwell *et al.*, 1995; Tait *et al.*, 2003; Higgins *et al.*, 2005). The Australian fish provinces of Unmack (2001) exemplify biogeographic regionalisation for aquatic biota. The provinces were derived from regional groupings of the AWRC drainage basins (Australian Water Resources Council, 1976) guided by concordance among the distribution patterns of obligate freshwater fish species. Tait *et al.* (2003) proposed that they delineate the highest level "macro-regions" of an Interim Freshwater Biogeographic Regionalisation for Australia (IFBRA). Lower levels within the proposed IFBRA hierarchy would be based on drainage network position (Tait *et al.*, 2003). However, the fish provinces will need revision to reflect the improved delineation of basin divides (Chapter 5). The somewhat arbitrary nature of province boundaries in poorly sampled, inaccessible regions (Unmack, 2001) is also cause for review.

New phylogenetic approaches offer the prospect of improved bioregionalisations for other faunal groups (e.g. turtles, Georges *et al.*, 2001), free of the confounding influences of stochastic processes (Tait *et al.*, 2003). It is not yet clear, however, how factors such as geographical scale and biogeographical histories affect taxonomic surrogacy (Margules and Pressey, 2000). Thus classifications based on just one or two taxonomic groups (e.g. fish and macroinvertebrates) may not be suitable surrogates for other aquatic taxa (S. Halse pers. comm. in Pressey and Adam, 1995; Butcher *et al.*, 2001; Paavola *et al.*, 2003). Bioregionalisations might better be based on ecologically meaningful subsets of taxonomic groups, reflecting the different determinants of distribution patterns among such groups (Williams and Hero, 2001).

Finer scale biological classification of rivers is limited by the lack of comprehensive inventory of Australian rivers and dependent ecosystems (Nevill and Phillips, 2004) and poor taxonomic knowledge of freshwater biodiversity (Georges and Cottingham, 2002). Comprehensive surveys (e.g. multiple taxonomic groups from instream, riparian and floodplain environments) are rare. Existing survey data are often biased towards easily accessible locations or larger, permanent streams (Williams, 1988; Kingsford, 1995). Even the collections of aquatic macroinvertebrates for the National River Health Program (NRHP) (Smith *et al.*, 1999; Turak *et al.*, 1999; Read, 2001) concentrate on areas of greatest management need and include few temporary streams, especially in arid regions. Low levels of taxonomic resolution (e.g. family) characteristic of many macroinvertebrate datasets, including the NRHP, may be inadequate for useful generalizations (Hawkins and Vinson, 2000; Tait *et al.*, 2003). Yet even with a large investment in inventory two important constraints on biological classification remain. Firstly, it is not

always possible to exclude the effect of anthropogenic activities on the distribution patterns of biota (O'Keefe and Uys, 2000). Translocations of native fish species, for example, by deliberate stocking or accidental release (e.g. aquarium releases or escapees from fish farms) are widespread (Burrows, 2002). Secondly, if long time series are not available, periodic population fluctuations can be incorrectly attributed to geographic variation (Frissell *et al.*, 2001). Such fluctuations are extreme in many floodplain wetlands with shifting spatio-temporal mosaics of assemblages reflecting complex patterns of connectivity dependent on highly variable flooding (Puckridge *et al.*, 2000; Timms and Boulton, 2001; Sheldon *et al.*, 2002; Reid *et al.*, 2003). Stochastic processes may be important determinants of community structure in intermittent or episodic streams. Two adjacent pools in a temporary river may have an entirely different faunal assemblage dependent upon the succession of colonizer species (Roux *et al.*, 2002). Classifications based on biotic community composition are essentially dynamic snapshots of ecosystems patterns that may need to be updated periodically (Jensen, Christensen *et al.*, 2001).

6.3. Habitat classification

Direct measures or descriptions of stream geomorphic characteristics are the basis of many of the classification schemes for rivers (Rosgen, 1994, 1996; Montgomery and Buffington, 1997; Brierley and Fryirs, 2000; Heritage *et al.*, 2001; Whittington *et al.*, 2001). These classifications have been used to:

- assess river condition (Fryirs and Brierley, 2000);
- manage environmental flows (Calvert *et al.*, 2001; Thoms and Sheldon, 2002);
- provide a framework for reporting (Whittington *et al.*, 2001) and ecological inventory (Savery *et al.*, 2001);
- define targets for river restoration and rehabilitation (Rosgen, 1994) and
- assist river management (Outhet *et al.*, 2001).

River Styles is a generic, non-prescriptive approach for evaluating the geomorphic character of river systems (Brierley and Fryirs, 2002). It has been applied state-wide in New South Wales (Outhet *et al.*, 2001) and in selected catchments in Tasmania, Queensland and South Australia (Brierley and Fryirs, 2002). River Styles are relatively homogenous reaches identified on the basis of river planform, geomorphic unit assemblages and bed material texture – the specific differentiating criteria dependent upon the valley setting (Brierley and Fryirs, 2000). Although River Styles recognises a hierarchy of nested spatial scales (catchment, landscape unit, reach, geomorphic unit and hydraulic (habitat) unit) the classification is not a nested hierarchy - a River Style might occur in more than one landscape unit (Thomson *et al.*, 2001; Jerie *et al.*, 2003). Accredited training programs, trademarking (Brierley and Fryirs, 2002) and the adoption

of a standard set of rules for identifying and labelling styles (Outhet *et al.*, 2001) improve the consistency of River Styles applications. Nevertheless, inconsistent interpretation and labelling of River Styles between catchments and inadequate consideration of the wider landscape and geomorphic history are a concern (Jerie *et al.*, 2003). While the River Styles approach emphasises the catchment and evolutionary context of river behaviour (Brierley and Fryirs, 2000), with limited time or inexperienced observers, applications tend to be more mechanical, based predominantly on visual assessment of the river planform and valley confinement (Jerie *et al.*, 2003).

Geomorphically defined reach types are also the foundation of the National River Classification System for Australian rivers proposed by Calvert *et al.* (2001) for environmental flow management. The top down hierarchical scheme places channel reaches into one of fifty *a priori* types that are nested within higher levels defined successively by climatic, geomorphic region, flow regulation and valley setting criteria. Notably, the scheme covers the full length of rivers from source to the ocean and recognises a number of estuarine and marine reach types. A list of geomorphic attributes is presented, but no differentiating criteria. However, establishing consistently interpreted criteria is likely to be problematic. The difficulties that may arise are evident in the classification of the Snowy River below Jindabyne Dam (Calvert *et al.*, 2001). Although bedrock and alluvial reach types are presented as uniquely nested within bedrock and alluvial valley segments respectively (figure 4 on page 20 in Calvert *et al.*, 2001), classified reaches are labelled as mixed types e.g. "Bedrock Confined Channel within a Bedrock/ Alluvial valley" setting.

The Rosgen stream classification system (Rosgen, 1994, 1996) is used extensively for inventory and assessment of streams in the United States (Jensen, Goodman, Frissell *et al.*, 2001; Savery *et al.*, 2001), particularly by the USDA Forest Service (USDA Forest Service Stream Systems Technology Center, 2001). It specifies quantitative delineative criteria for measured attributes of channel form (entrenchment, channel planform, width:depth ratio, sinuosity, slope and dominant substrate) to classify reaches. Although quantitative, the criteria may not be applied consistently because of unreliable methods for identifying and measuring attributes such as bankfull width (Savery *et al.*, 2001) and bias associated with unrepresentative field measurements (Myers and Swanson, 1997). The prescriptive classes may be region specific and are not applicable to unique Australian river forms (Calvert *et al.*, 2001; Brierley and Fryirs, 2002). Rivers that don't conform could be forced into one of the predefined types, with potentially disastrous consequences for subsequent management responses (Malakoff, 2004, G.Brierley, Macquarie University pers. comm. Nov. 2002) or classes must be added or modified (e.g. Savery *et al.*, 2001). The geomorphic significance of the delineative criteria is also questionable (Miller and Ritter, 1996). In fact, Goodwin (1999) criticises classifications focussed on channel form generally for a lack of process understanding. As he warns, channel

forms are just the end products of a complex dynamic system and may be non-unique manifestations of underlying controlling factors and processes. He concludes, therefore, that natural kinds of rivers are more likely to be discovered by measuring fluvial processes or controlling factors.

A more process-based classification framework is introduced by considering sediment transport processes (Nanson and Croke, 1992; Whiting and Bradley, 1993; Nanson and Knighton, 1996; Montgomery and Buffington, 1997; Alabyan and Chalov, 1998). Such classifications are likely to be more effective for assessing channel condition, management response and relations to ecological processes (Montgomery and Buffington, 1997) but the process threshold values employed may not be uniformly relevant across large areas (Jerie *et al.*, 2003).

Whittington *et al.* (2001) mapped Functional Process Zones (FPZ) for the major rivers of the Murray Darling Basin. FPZs are lengths of river with similar discharge and sediment regimes, defined by gradient, stream power, valley dimensions and boundary material. They are nested within three valley process zones (source, transport and deposition) broadly identified by their sediment transport characteristics. The classification is used to stratify river valleys for reporting of the Murray Darling Basin Sustainable Rivers Audit (Whittington *et al.*, 2001) and supports an ecosystem approach for determining environmental water allocations (Thoms and Sheldon, 2002). Its value is enhanced by the accompanying conceptual models of river function that relate physical factors within each FPZ to ecosystem structures and processes (Thoms *et al.*, 2001).

The mapping of geomorphologic classifications, including FPZ's, depends on expert interpretation of data collected from field survey, complemented by remotely sensed imagery, aerial photographs, and historical data. Field observations must be sufficiently representative of a reach to ensure unbiased type assignments (Brierley and Fryirs, 2002). These information requirements restrict the number of streams that are classified. A full River Styles assessment for instance, may take up to a day per reach (Brierley and Fryirs, 2002). Alternatively, the probability of occurrence of geomorphic types may be modelled using larger scale landscape variables at reach (Young, Ogden *et al.*, 2001) and geomorphic unit (habitat) scales (Jeffers, 1998; Davies *et al.*, 2000). New techniques for terrain analysis (Wilson and Gallant, 2000; Dowling *et al.*, 2003; Gallant and Dowling, 2003) and sediment modelling (Prosser *et al.*, 2001) enable improved representation of critical landscape variables. Modelling geomorphic process types using temporally stable landscape attributes may overcome another difficulty of classifications based on channel form – the low temporal stability of classified forms (< 10 years) (Naiman *et al.*, 1992). Predicted types also provide a reference state for assessing current condition (Davies *et al.*, 2000), though model development and mapping will be constrained by the availability of site data, suitably representative and unaffected by human activities, and high resolution terrain and substrate spatial data.

The structural features that are the basis of geomorphological classifications are also the physical elements of the habitat of biota and so may be important determinants of biological variation (Maddock, 1999; Newson and Newson, 2000). Geomorphological diversity contributes to the complexity of riverine habitats (Thoms and Sheldon, 2002). However, the geomorphologist's view of the criteria that differentiate stream channel types may not be the same as those to which the biota respond (Karr and Chu, 1999). For instance, Thomson *et al.* (2004) found River Styles did not provide a good representation of macroinvertebrate distribution patterns even within a single bioregion. Moreover, geomorphic elements are only one of the physico-chemical determinants of ecological habitat (Thomson *et al.*, 2004). The interaction of geomorphology and hydrology, for example, produce a distinct mosaic of hydraulic features (depths, velocities, shear stresses) (Maddock, 1999).

Semeniuk and Semeniuk (1995) recognize a higher-level interaction of geomorphology and hydrology in the primary delineative criteria they adopt for generic wetland classification (Semeniuk and Semeniuk, 1997). The classification defines 13 primary classes from the combination of five broad landform types (basin, channel, flat, slope, highland) and 4 classes of water permanence. Vegetation cover (spatial arrangement and internal organisation of vegetation assemblages), salinity and morphology (planform and shape) are secondary classification descriptors for differentiating riverine types. The classification ignores the functional linkages between the river and its floodplain by separating floodplain and channel types at the highest level. Although this separation may be justified for some wetlands on old floodplains controlled by groundwater rather than fluvial processes (Cowardin *et al.*, 1979), this is certainly not the case for most wetlands in Australia (Walker, 1993; Kingsford *et al.*, 2004). More useful for conservation applications is Semeniuk's regional level classification of wetlands into suites of similar or related types ("consanguineous suites") (Semeniuk, 1988). These assemblages are more likely to represent functional ecological units. The classification has been used in regional studies in WA (Nevill and Phillips, 2004) and the Northern Territory, but does not provide a workable basis for a national classification of Australian river systems. Seldom is it feasible to collect the data necessary for application of the secondary descriptors over large areas (Kingsford *et al.*, 2004) and despite its intended global applicability (Semeniuk and Semeniuk, 1997), Begg *et al.* (2001) found it necessary to add a number of types and use locally recognised terms to classify wetlands in the Daly River Basin.

Similar concerns also apply to national application of the classification system of Blackman *et al.* (1992). Like many other wetland classification systems, they adopt the broader RAMSAR definition of wetlands (Ramsar Convention Secretariat, 2003), recognising riverine as one of five major wetland systems together with estuarine, lacustrine, palustrine, and marine. Riverine systems include only those wetlands flowing within an open channel. Secondary criteria of hydroperiod, substrate and dominance (flora or fauna) type are then used to subdivide these

major systems with modifiers describing the water regime, soil, water chemistry and anthropogenic influence. The modifying criteria refer to average conditions of dynamic wetland properties and therefore require extended time series of observations for application. The system uses field description of representative sites from assemblages of wetlands delineated on remotely sensed images. A biogeographic regionalisation and geomorphic land types provide the stratification for selection of survey sites. While not providing an exhaustive inventory of all wetlands, the system does offer an operational framework for systematic, regional scale assessment of wetland systems (Blackman *et al.*, 1995).

This classification system, based on the U.S. hierarchical wetland classification of Cowardin *et al.* (1979), has been used in regional studies in Queensland (Nevill and Phillips, 2004). Characteristic associations of wetland types and geomorphic land types within wetland aggregations form “regional wetland habitats” that could be a useful basis for conservation planning for wetland biodiversity (Blackman *et al.*, 2002). However, Phillips *et al.* (2002) expresses concern about the sensitivity of the classification to the order in which the differentiating criteria are applied. Blackman *et al.* (1992) recognise that at lower levels the system is open-ended and incomplete, a recurring issue for all classifications that rely on prescriptive, *a priori* categories. The ecological significance of some class divisions may also need review. For example, the class “unconsolidated bottom” includes all wetlands with at least 25% cover of particles smaller than stones, and a vegetative cover less than 30%” (Blackman *et al.*, 1992) but it is unlikely that these values, derived from the U.S. system (Cowardin *et al.*, 1979), are universally appropriate. Still, field survey data are maintained in a spatially referenced database, the Wetland Information System (Blackman *et al.*, 1995), allowing flexible access and analysis and alternative classification. The database is also linked to remote sensing and other natural resource, cadastral and cultural information providing a powerful tool to support a range of conservation applications. The classification is not applicable nationally but the system may be a model for wetland mapping and inventory and development of a national wetland information system for *a posteriori* classification.

Like many generic wetland classifications, the classification uses simple, broad categories of hydroperiod to characterise flow. But hydroperiod is just one of the ecologically important components of flow (Poff *et al.*, 1997; Hart and Finelli, 1999) operating over multiple temporal scales (Puckridge *et al.*, 1998; Thoms and Parsons, 2003) that are used to classify rivers. Puckridge *et al.* (1998), for example, used hydrological measures describing a number of facets of flow variability to classify global rivers. Hughes and James (1989) derived five hydrologically distinct regions from a classification of Victorian stream gauges using flow statistics for annual, monthly and low flows and peak (daily) discharges. Such classifications, however, are only possible for a small sub-set of gauged streams. Methods to extend flow parameters to ungauged streams are developing (see review in Croke, 2002), although

confounding the detection of natural spatial variation are the effects of flow regulation and catchment development. Also still developing is an understanding of which of the multitude of hydrological indices that may be used to characterise flow (e.g. Puckridge *et al.*, 1998; Groun and Marsh, 2000; Pegg and Pierce, 2002) represent ecologically meaningful variation (Olden and Poff, 2003).

6.4. Landscape classification

Landscape or environment classifications are primarily based on meso-scale attributes of the physical environment, including geology, climate, topography and vegetation types. Ecoregions are landscape regionalisations within which the mosaic of ecosystem components (biotic and abiotic, terrestrial and aquatic) is relatively homogeneous and different from adjacent regions (Omernik and Bailey, 1997). They support an ecosystem based approach to natural resource planning and management in many countries (Clarke *et al.*, 1991; Warry and Hanau, 1993; Omernik, 1995; Bailey, 1996; Bryce and Clarke, 1996; Uhlig and Jordan, 1996; Harding and Winterbourn, 1997; Wells and Newall, 1997; Sandin and Johnson, 2000).

The Interim Biogeographical Classification of Australia (IBRA), for example, is a continent-wide regionalisation of landscape patterns, inferred from regional and continental scale data on climate, geomorphology, landform, lithology and expert knowledge of characteristic flora and fauna (Thackway and Cresswell, 1995). IBRA was developed to support the National Reserves System program (Pigram and Sundell, 1997) but may not adequately represent variation in the biotic composition of rivers (Wells and Newall, 1997; Marchant *et al.*, 1999; Turak *et al.*, 1999). Hawkins *et al.* (2000), synthesising a series of international studies, similarly concluded that broad landscape regionalisations were unable to effectively partition biotic composition (algal, invertebrate and fish) among sites. The generally weak performance of ecoregions has been attributed to poor representation of longitudinal gradients in rivers (Snelder and Biggs, 2002) and dispersal barriers associated with drainage divides (Tait *et al.*, 2003), and to local habitat variation within a region (Hawkins *et al.*, 2000). In contrast, terrestrial ecoregions with low within-region altitudinal variation and boundaries that usually followed major catchment divides showed “remarkably good concordance” with benthic invertebrate assemblages (Rabeni and Doisy, 2000). Accordingly, the IBRA sub-regions (Morgan, 2001) that divide major geomorphic units may perform better, particularly if completely aquatic organisms are considered separately from those with terrestrial life stages or distributional abilities (Tait *et al.*, 2003).

Inconsistent application and poor communication of the decision rules for IBRA’s boundary placement remains a weakness. The individual State and Territory regionalisations that form IBRA were derived using different combinations of biological and/or environmental data, covering a range of spatial scales and classification approaches. Methodological variations are

most apparent in the IBRA sub-regions along state borders (Environment Australia, 2000). For example, Victoria utilised detailed vegetation mapping to delineate regions, whereas New South Wales and South Australia used coarser mapping of land systems and geological and geomorphological features to define boundaries (Environment Australia, 2000), confounding the interpretation of cross-border regions. IBRA regions for instance, may cut across the boundaries of several quite different climatic regions (Landsberg and Kesteven, 2002; Hutchinson *et al.*, 2005).

IBRA uses several variably applied factors to define individual ecoregions. In contrast, the River Environment Classification (REC) of Snelder and Biggs(2002) employs the “top down” hierarchical controlling factor method of ecoregion definition (Bailey, 1996), recognising environmental factors control ecological variation at characteristic scales (Figure 6.2). At the coarsest scale, river segments are classified into six broad climate categories that characterise the dominant climate of the upstream catchment. These climate classes are successively subdivided using prescriptive categories of source of flow, geology, land cover, network position and valley landform. Unlike other ecoregion classifications, REC classifies river segments individually according to the differentiating criteria, producing geographically dispersed classes. It recognizes however, the influence of the upstream catchment and therefore better represents the longitudinal gradients in aquatic ecosystem properties (Snelder and Biggs, 2002). The REC approach has been used to classify all the rivers of New Zealand at a mapping scale of 1:50,000 (Snelder and Biggs, 2002), providing a useful framework for water allocation (Snelder *et al.*, 2001), identifying reference and impaired reaches for stream eutrophication (Biggs *et al.*, 2002) and understanding spatial variation in hydrologic characteristics (Snelder *et al.*, 2005). However, there are some potential problems in applying the REC approach to Australian rivers. Within the first four system levels, the REC category is determined by catchment average or for categorical data (e.g.geology) dominant values, but these summary values may be meaningless, particularly in large catchments, and not reflective of the integrated effect of controlling factors. The quantitative, prescriptive classes enable the classification to be applied consistently and explicitly but different categories would need to be developed for applications outside of New Zealand (Snelder and Biggs, 2002). For example, the majority of Australian rivers would fall within just one of REC’s climate classes, the “dry” category for rivers with a catchment mean annual effective precipitation of 500mm or less. However, little data are available to set ecologically relevant class boundary values. Furthermore, these values are likely to vary because the actual character of the stream is a function of a complex interaction of environmental variables (Omernik, 1995; Jerie *et al.*, 2003).

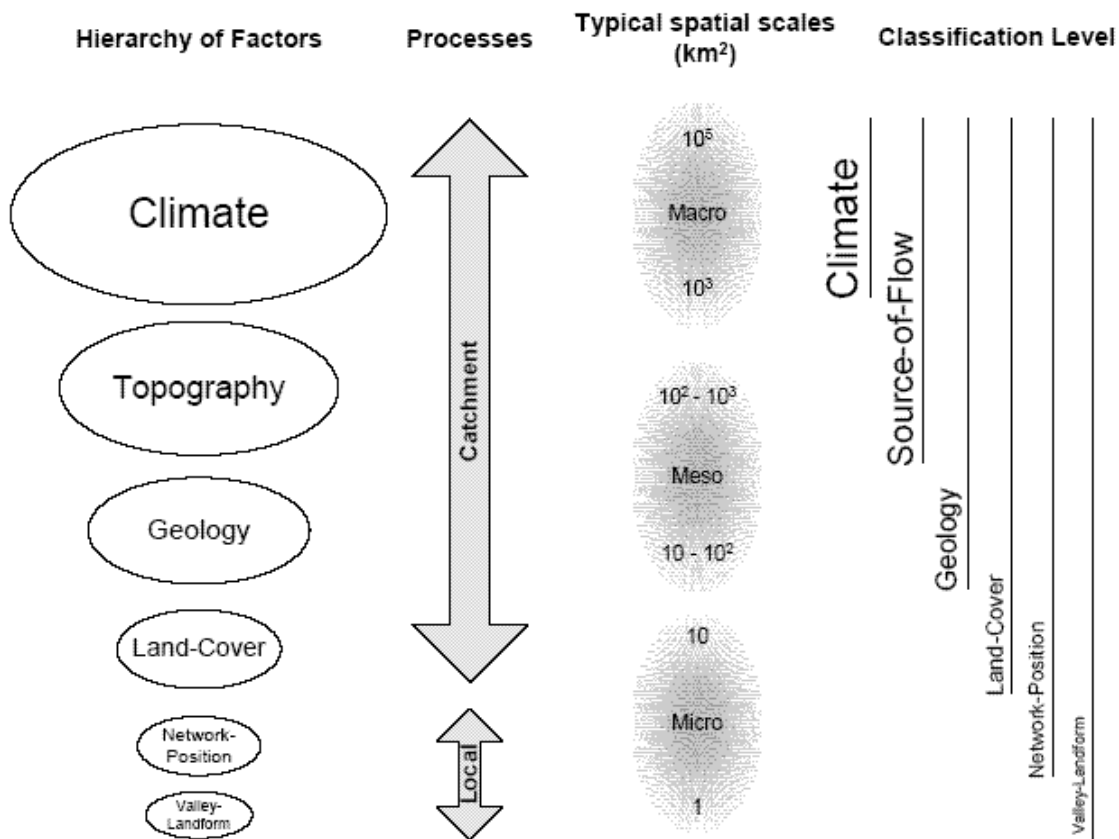


Figure 6.2 The River Environment Classification (REC) – controlling factors and characteristic spatial scales (after Snelder, Biggs *et al.*, 2004)

Numerical procedures, such as clustering, provide an alternative approach to landscape classification. Rivers are grouped on the basis of their shared similarities across a range of attributes. The number of groups may be chosen to suit specific objectives and the relationship between them quantified. Phillips *et al.* (2002) compared intuitive and numerical agglomerative classification procedures in the Burnett River catchment in Queensland. The types (“Bioregional Aquatic Systems”) derived using numerical procedures were preferred to the a priori intuitive classifications produced by an expert panel. However, it was necessary to combine field surveyed attributes of channel form and dominant substrate with the desktop derived landscape attributes (annual mean rainfall, slope, catchment area and distance to the sea) to delineate ecologically meaningful types (Phillips *et al.*, 2002). This need not be the case if landscape attributes are chosen to more comprehensively represent the range of controlling factors that drive stream ecosystem processes (e.g. Jerie *et al.*, 2003).

The definition of classes numerically as an emergent property of the primary data is one of the features that distinguish environmental domain analysis from other landscape classification approaches. Environmental domains are spatial units, for which attributes of meso-scale climate, substrate (regolith and soils) and topography are relatively homogenous at a prescribed level of dissimilarity (Figure 6.3). These attributes integrate the effects of those primary environmental

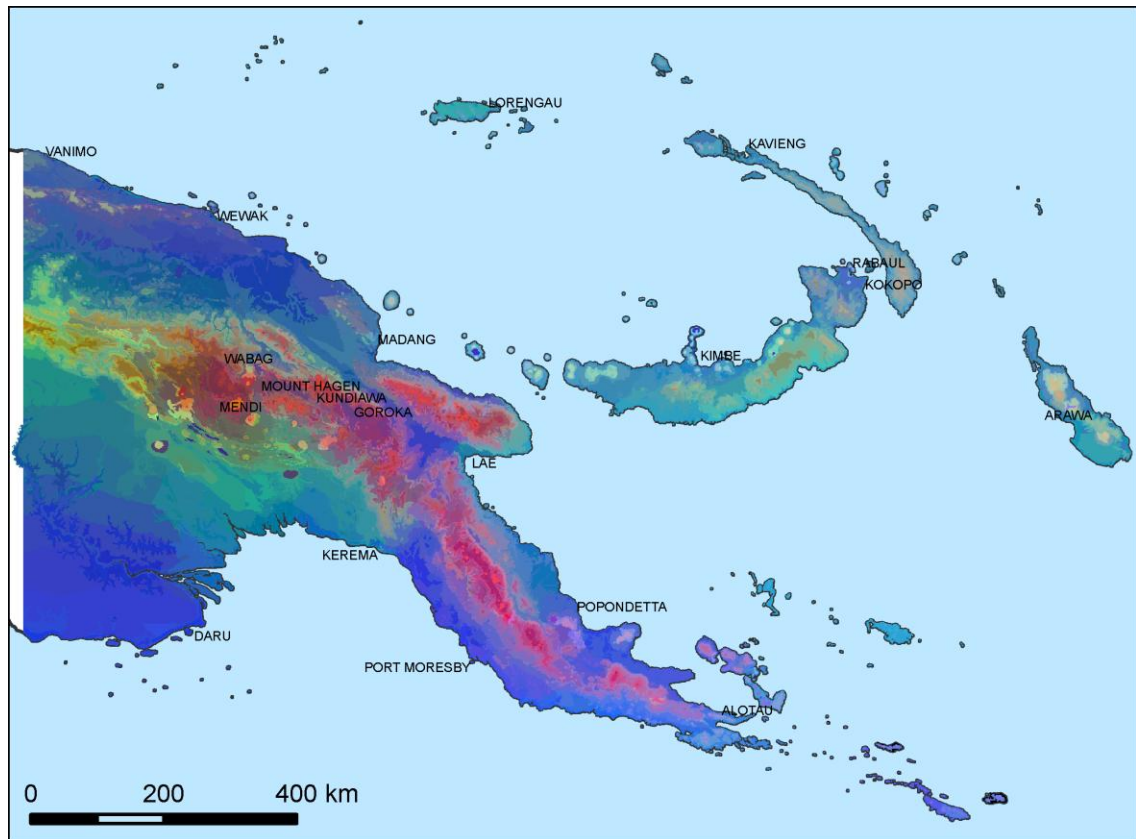


Figure 6.3 Environmental domains of Papua New Guinea used as biodiversity surrogates to identify priority areas for conservation (after Nix *et al.*, 2000).

variables (e.g. light, mineral nutrients, moisture, temperature) that drive landscape physical processes and biological function and so provide the basis for the definition of ecosystem patterns (Mackey *et al.*, 1988; Mackey *et al.*, 1989; Nix, 1992, 1997; Mackey *et al.*, 2001).

Environmental domain analysis also differs from ecoregion approaches in two other important respects. Firstly, classes are delineated in environmental rather than geographic space and secondly, hierarchical levels are defined using an agglomerative “bottom up” approach. The environmental domain approach focuses on collating primary attribute data at highest possible spatial resolutions, rather than producing a single generic classification, so that by varying the numbers of groups, variable weightings, or choice of clustering strategy, a classification may be tailored to meet different objectives. Environmental domains have been used widely to represent patterns of terrestrial biodiversity (Mackey *et al.*, 1989; Lewis *et al.*, 1991; Belbin, 1993a; Kirkpatrick and Brown, 1994; Mackey *et al.*, 2000; Nix *et al.*, 2000; Mackey *et al.*, 2001; Leathwick *et al.*, 2003; Trakhtenbrot and Kadmon, 2005) and to assess the representativeness of biological survey data (Richards *et al.*, 1990; Mackey *et al.*, 2001) but were only recently developed for river systems (Jerie *et al.*, 2003). Environmental domains, based on variables selected to represent the important controls on stream development (climate, topography, lithostructure and process history), effectively accounted for regional variation in the character and behaviour of Tasmanian rivers (Jerie *et al.*, 2003).

6.5. Conclusions

Different approaches to river classification were reviewed. There is no universal system for classifying streams, stream habitats or their biotic communities (Jensen *et al.*, 2001b). None provide a sufficient basis for all of the conservation tasks for which a classification is needed.

Landscape classifications such as environmental domains or REC, utilise existing, geographically referenced sources of data and an automated spatial analysis framework, making it feasible to classify all streams, both large and small. They address the spatial distribution of relatively consistent associations of environmental factors that drive the pattern of flow, channel morphology, substratum, temperature and mineral nutrients that collectively define the physical habitat template of stream ecosystems. As a result, they produce temporally stable groupings of streams with similar response potential (i.e. range of possible states) regardless of current natural or anthropogenic disturbances (Bailey, 1996). Biogeographic classifications reflect the effects of historical processes that limit the pool of species within a river. Together, landscape and biogeographic classifications represent the range of ultimate controls on aquatic ecosystem patterns and processes and could support conservation applications at regional to national scales including assessment of ecological value, design of comprehensive surveys, reporting progress towards conservation targets and assisting co-operation and co-ordination among jurisdictions. They also provide the landscape scale context for the finer scale classifications, based on direct measures of stream ecological, hydrological and geomorphological characteristics that are necessary for catchment specific planning and management, assessment of current condition and design of appropriate targets for restoration or rehabilitation. These direct measures integrate the proximate factors that control aquatic ecosystems but would require a major investment in systematic and comprehensive inventory of rivers and associated ecosystems using nationally agreed survey protocols.

Chapter 7. Landscape classification of Australian rivers and streams

The review of river classification systems in Chapter 6 found that those based on direct measures of riverine biotic assemblages or channel geomorphic characteristics are essential for many management tasks but they are not yet feasible to implement continentally. Moreover, their low temporal stability makes them inappropriate for many conservation planning applications. On the other hand, biogeographic and landscape classifications present viable options for continent-wide classification and a suitable basis for many conservation planning tasks.

This chapter explores one of these options; developing and testing national landscape classifications for Australian rivers and streams. Landscape classification was focussed upon because suitable data were available for consistent attribution of hydrologic units at a range of spatial scales. Since no single landscape classification method will be appropriate for all applications, the classification requirements of continental scale conservation planning are considered first (Section 7.1). These set the context for decisions on the classificatory strategy (Section 7.2.1), including the objects of the classification, the attributes that characterize these objects and the classification method. The most important of the conservation planning needs, that the classification discriminate ecologically meaningful variation among streams, is considered in Chapter 8. Section 7.2.2 describes the application of this classificatory strategy, detailing the derivation of landscape attributes and clustering strategy.

The resulting classifications are presented in Section 7.3 with descriptions and evaluation (Section 7.3.1.) and hierarchical organization (Section 7.3.2). Choices of classificatory strategy can have a profound influence on outcomes. Section 7.3.3 explores the appropriateness of one such decision, specification of the number of clusters. The chapter concludes (Section 7.4) with a discussion of the suitability of the landscape classifications for conservation planning, their major limitations and opportunities for improvement.

7.1. Guiding principles

There are numerous approaches to classify rivers (Chapter 6). To select the most appropriate strategy consideration of the conservation planning requirements of a classification is necessary (Box 7.1).

Box 7.1 Characteristics of a classification for systematic conservation planning for rivers and streams
1. Discriminates ecologically meaningful variation at multiple spatial scales 2. Recognizes connectivity within and barriers between drainage networks 3. Temporally stable groupings (>100yrs) 4. Derived using explicit and repeatable methods 5. Applicable to both large and small streams 6. Able to be mapped continentally for all planning units

It is important that the classification embrace the whole channel network (Montgomery and Buffington, 1997) at a number of spatial scales, realizing that no single scale will be appropriate for all purposes (Jensen *et al.*, 1996). Classification of whole streams or drainage basins will not capture longitudinal variation in stream ecological and geomorphological characteristics (Naiman *et al.*, 1992) yet it is usually impractical to classify and map individual features at very fine scales (e.g. micro-habitat) for other than small areas. A multi scale classification recognizes the range of scales at which ecological processes operate (Frissell *et al.*, 1986; Naiman *et al.*, 1992; Snelder and Biggs, 2002) and conservation evaluation (Austin and Margules, 1986) and management responses take place.

A process-based understanding of spatial linkages within a catchment is necessary for effective classification (Montgomery and Buffington, 1997). Patterns of connectivity within and between catchments often regulate the distribution of aquatic biota (Frissell *et al.*, 2001) and the transfer of energy and materials (Piégay and Schumm, 2003). Obligate freshwater biota (e.g. freshwater fishes) experience significant biogeographic constraints (Unmack 2001). Ordinarily restricted by existing catchment divides, their distribution also reflects changes to hydrologic connectivity linked to past infrequent events such as drainage rearrangements or sea level changes (Unmack 2001). Tait and his colleagues (2003) believe that recognition of drainage divides and patterns of connectivity, both current and historic, is a key consideration for the delineation of freshwater bioregions.

A classification needs to discriminate ecologically meaningful variation in ecosystem processes and reflect the distribution patterns of aquatic biota (Phillips *et al.*, 2002). While this is often assumed it also needs to be explicitly stated and tested (Bourgeron *et al.*, 2001d). The requirements for data and specialist knowledge should be commensurate with the need to map the classification for all planning units. Differentiating criteria must be consistently and transparently applied for valid comparisons between potential conservation areas and to help ensure that resulting conservation actions are defensible (Austin and Margules, 1986; Margules and Pressey, 2000). Failure to clearly explain boundary decisions could lead to frequent

revisions and poor acceptance and use of the classification (Perera *et al.*, 1996; Hargrove and Hoffman, 1999).

Classifications are better based on attributes that are temporally stable or that effectively integrate temporal variation (Bennett *et al.*, 2002) rather than dynamic characteristics that describe the current state. Evaluating potential states rather than current condition will assist in distinguishing natural variability from human disturbance (Naiman *et al.*, 1992). Inclusion of anthropogenic effects with natural variation could produce arbitrary groupings that reflect neither natural affinities nor classes of degradation (O'Keefe and Uys, 2000).

7.2. Methods

7.2.1. Classificatory strategy

The desirable characteristics of a classification outlined in Box 7.1 guided the choice of a classificatory strategy for Australian rivers and streams, in particular, the clustering method, objects, attributes, and number of groups.

Units from the spatial framework (Chapter 4) are logical entities for classification. They respect the catchment constraints and natural hierarchical structure of the stream system. Stream segments, the smallest unit in the framework, were adopted as the objects of the classification. Larger catchment units are likely to be heterogenous for key landscape variables and therefore more difficult to characterise for classification.

The classes may be defined *a priori* by specifying the boundaries between them as the outer limits of the features that characterize them (O'Keefe and Uys, 2000) (e.g. New Zealand's River Environment Classification, Snelder and Biggs, 2002) or generated as an emergent property of the primary data based on the shared similarities of key attributes (e.g. environmental domains Mackey *et al.*, 1988). Defining class boundaries *a priori* assumes all possible types are already known and relies on expert knowledge (Phillips *et al.*, 2002). However, there is little information on ecologically relevant thresholds for key landscape attributes that would provide the basis for *a priori* definition of categories. Hence, this study used numerical clustering procedures to derive classes directly from the data.

Numerical clustering procedures integrate many attributes in a consistent and repeatable manner (Austin and Margules, 1986) and are less likely to be influenced by personal biases or prominent features (Thandaveswara and Sajikumar, 2000; Phillips *et al.*, 2002), although groups may be more difficult to interpret (Mosley, 1987). While subjective choices remain (e.g. attributes, classificatory algorithm and number of groups) these are explicit and therefore transparent and easily modified (Austin and Margules, 1986; Nix, 1992). They can be adapted to work effectively with different data types (categorical and numerical) (Bao, 2000) and

because the relationships within and between groups are quantified, numerical clustering allows application of formal methods for evaluating representativeness (Austin and Margules, 1986; Belbin, 1993a).

Some measure of association or distance between objects is fundamental to most clustering methods. Hierarchical methods, either agglomerative or divisive, require a matrix of (dis)similarity values between every pair of objects. For N objects there are $N(N-1)/2$ pairs excluding the self-similarities. Non-hierarchical clustering methods only compute similarity values between an object and a group centroid, dramatically reducing the computational requirements for very large numbers of objects. This study uses the non-hierarchical clustering strategy, ALOB from the PATN software package version 3.6 (Belbin, 1993b) operating on a Sun Solaris computer system. With nearly 1 million stream segments to classify, hierarchical methods were not a viable option. Even if it were computationally feasible, it would be almost impossible to interpret a hierarchical classification of so many objects - the dendrogram that displays the relationship between the objects would extend over hundreds of pages! Furthermore, ALOB produces results that are at least comparable with traditional agglomerative hierarchical methods (Belbin, 1987; Belbin and McDonald, 1993). It is commonly used in environmental domain analysis where there is a need to classify very large numbers of objects (e.g. grid cells or sub-catchments) (Mackey *et al.*, 1988; Mackey *et al.*, 1989; Lewis *et al.*, 1991; Nix, Stein and Stein, 1992; Nix, Stein, Stein *et al.*, 1992; Nix *et al.*, 2000).

ALOB groups the objects (i.e. stream segments) according to the similarity of their attribute values. The set of landscape attributes selected was informed by a review of the landscape attributes used for river classifications elsewhere (Table 7.1) and understanding of the key landscape drivers of river ecosystems obtained from the literature. The final selection of attributes was constrained by the need for continental coverage, limiting choices to those attributes for which suitable spatial data were available.

Table 7.1 Landscape attributes used to classify rivers

Landscape Factor	Attribute	References
Geology	Type (catchment, channel floor)	Frissell <i>et al.</i> , 1986; Huang and Ferng, 1990; Jensen, Goodman, Bourgeron <i>et al.</i> , 2001; Jerie <i>et al.</i> , 2001; Roux <i>et al.</i> , 2002; Snelder and Biggs, 2002; Sowa <i>et al.</i> , 2004
Process history	Areas affected by aeolian, karst, glacial, periglacial and fluvial processes	Jerie <i>et al.</i> , 2001
Climate	Rainfall (annual and monthly average; 24 hour maximum storm intensity; annual average daily maximum, variability (Cv); seasonality); Potential evapotranspiration, Effective precipitation; Air temperature (annual and monthly average)	Fatchen and Lustig, 1986; Frissell <i>et al.</i> , 1986; Huang and Ferng, 1990; Wells and Newall, 1997; Harris <i>et al.</i> , 2000; Calvert <i>et al.</i> , 2001; Jensen, Goodman, Bourgeron <i>et al.</i> , 2001; Phillips <i>et al.</i> , 2002; Roux <i>et al.</i> , 2002; Snelder and Biggs, 2002; Jerie <i>et al.</i> , 2003; Sowa <i>et al.</i> , 2004
Topography	Catchment relief; Catchment relief ratio; Valley slope; Catchment shape (Elongation ratio); Catchment area; Valley side slopes (Length, Angle); Valley confinement (valley width, percent channel abutting valley margin, cross sectional terrain curvature); Network position (stream order; distance from source); Slope break (profile curvature); Altitude	Strahler, 1957; Frissell <i>et al.</i> , 1986; Huang and Ferng, 1990; Whiting and Bradley, 1993; Wells and Newall, 1997; Jensen, Goodman, Bourgeron <i>et al.</i> , 2001; Jerie <i>et al.</i> , 2001; Phillips <i>et al.</i> , 2002; Roux <i>et al.</i> , 2002; Snelder and Biggs, 2002; Sowa <i>et al.</i> , 2004
Soils	Type	Fatchen and Lustig, 1986; Frissell <i>et al.</i> , 1986; Huang and Ferng, 1990; Sowa <i>et al.</i> , 2004
Vegetation	Potential natural type	Wells and Newall, 1997; Roux <i>et al.</i> , 2002
Landuse	Category; Percent cleared	Frissell <i>et al.</i> , 1986; Huang and Ferng, 1990; Snelder and Biggs, 2002

7.2.2. Classifying Australian rivers and streams

Figure 7.1 sets out the major steps followed to classify Australian rivers and streams. The necessary spatial data were compiled from data sources identified as the best of those available with consistent continent-wide coverage. Gridded data layers were derived from these data for each of the selected landscape attributes using a combination of specially written procedures and existing programs. The grid data structure is used for compatibility with the spatial framework. It is also an efficient format for storing and analysing large amounts of data. Stream segments were then attributed as appropriate with either the value of the attribute at the pour-point cell or a summary statistic derived from the values of the grid cells comprising the stream segment and/ or its direct sub-catchment, computed using the Arc/Info Grid ZONAL functions. Where it was necessary to route or accumulate cell values through the drainage network to derive attribute values (e.g catchment area, catchment water balance, relief ratio), specially written FORTRAN programs were used (Section 4.2.1.4). In this way, a data matrix of stream segments by landscape attributes was formed as input to the ALOB numerical clustering program.

ALOB was used to classify stream segments according to their shared similarity across all of the landscape attributes combined. A second classification was also produced by hierarchically combining separate groupings based on attributes of climate, terrain and a modelled catchment water balance (“flow”) respectively, and existing classifications of lithology and pre-European vegetation. Evaluation of the two resulting classifications marks the final stage of classification development. This includes interpretation of groups and assessment of their ecological relevance. The latter evaluation task is left until Chapter 8. A detailed description of each of the steps set out in Figure 7.1 follows.

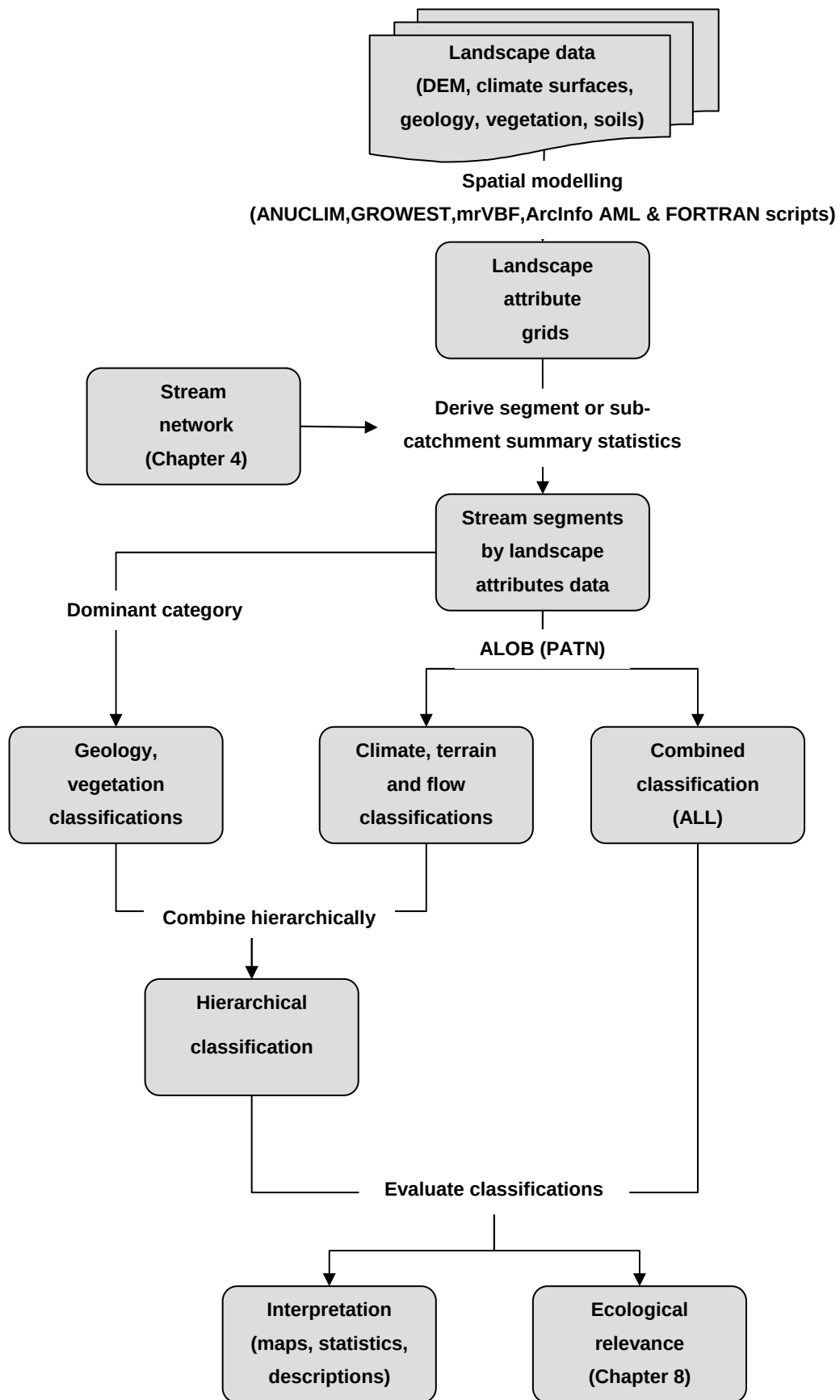


Figure 7.1 Developing a landscape classification for Australian rivers and streams.

7.2.2.1. Attributing landscape characteristics

Climate

Climate ultimately controls many of the processes that shape stream ecosystems (Knighton, 1998). Climate imposes fundamental constraints on biotic distribution patterns (Heino, 2001). Solar radiation, for example, is the major factor influencing stream temperature (Johnson, 2003) and through it the rate of in-stream chemical and biological processes. Rainfall and temperature affect rates of weathering of rock and hence the release and transport of solutes and bed and bank materials (Ollier and Pain, 1996; Knighton, 1998; Nanson *et al.*, 2005). Rainwater may also contribute significant chemical inputs to the stream (Cushing and Allan, 2001). Climatic parameters supply surrogates for the critical environmental regimes (light, moisture, thermal) that control plant growth (Nix, 1981, 1992, 1997). In turn, plant productivity influences catchment erosion and runoff processes and organic matter inputs to streams.

Ten attributes were used to characterize the influence of climate on stream ecosystems (Table 7.2). Excluding rainfall intensity, these attributes were derived by summarizing grid cell estimates of climatic parameters computed with the BIOCLIM and GROCLIM programs from the ANUCLIM package (Hutchinson, Houlder *et al.*, 2000). The BIOCLIM program produces a set of 35 bioclimatic parameters summarising annual and seasonal mean conditions, extreme values and intra-year seasonality (Nix, 1986; Hutchinson, Houlder *et al.*, 2000), from weekly mean values of maximum temperature, minimum temperature, rainfall, radiation and evaporation. BIOCLIM interpolates the weekly mean climate values with cubic Bessel interpolation from long-term monthly mean estimates. It produces gridded estimates of the monthly mean values by coupling elevation dependent, thin plate smoothing spline surfaces to a DEM (Hutchinson, Houlder *et al.*, 2000). Many of the 35 bioclimatic parameters are strongly correlated within any given region, so a subset of seven was selected to represent the major facets of the climatic influence on stream ecosystem patterns and processes (Table 7.2). The gridded estimates were computed for each cell of the national 9 second DEM version 2.1 (Section 4.2.1.1).

GROCLIM is a simple, generalised model of plant growth (Hutchinson, Houlder *et al.*, 2000), adapted from the GROWEST program (Nix, 1981). Like BIOCLIM, GROCLIM will use the long-term monthly mean climate values estimated for each grid cell in a DEM, interpolated to weekly values. It derives separate indices, each scaled between zero (completely limiting to growth) and one (not limiting), that describe the plant response to the light, thermal and moisture regimes. The Growth Index (GI) is the product of these three indices. Optional parameters control the shape and range of the temperature response curve. GROCLIM supplies parameters for three default plant types (microtherm, mesotherm and megatherm), each with a

Table 7.1 Climatic attributes used to classify stream segments. Attributes were computed as the average of the values of the grid cells that comprise the stream segment and where indicated, those that fall within its sub-catchment.

Source	Method of attribution	
	Stream segment grid cell average	Sub-catchment grid cell average
BIOCLIM (Hutchinson, Houlder <i>et al.</i> , 2000)	Annual mean temperature (°C)	Annual mean rainfall (mm)
	Hottest month mean temperature (°C)	Driest quarter mean rainfall (mm)
	Coldest month mean temperature (°C)	Wettest quarter mean rainfall (mm)
		Annual mean radiation (MJ/m ² /day)
GROCLIM (Hutchinson, Houlder <i>et al.</i> , 2000)		Annual mean Growth Index (microtherm plants)
		Annual mean Growth Index (mesotherm plants)
		Annual mean Growth Index (megatherm plants)
NLWRA (National Land and Water Resources Audit, 2000)		Rainfall erosivity R factor (MJ mm)/(ha hr yr)

different optimal temperature range for growth. The moisture index is derived from a simple water balance calculation. It was computed for the default uniform soil, a clay loam soil with 150mm available soil water. It was not necessary to account for the spatial variability in soil properties because long-term mean water balance calculations can assume negligible changes in soil water storage. To reduce computation time the DEM was resampled to 36 seconds (about 1km) resolution with the Arc/Info GRID RESAMPLE function using bilinear interpolation. This scale is also closer to the optimum scale of elevation dependence of monthly precipitation of 5 to 10km (Hutchinson, 1998; Sharples *et al.*, 2005) than the original DEM resolution of 9 seconds. GI values were calculated independently at each grid cell for each plant type and stream segments attributed with the average sub-catchment values (Table 7.2).

Rainfall erosivity is the potential for rainfall induced soil loss. It is commonly described by the rainfall erosivity R factor in the revised universal soil loss equation (RUSLE) used to estimate sheet and rill erosion rates (e.g. Lu *et al.*, 2001). The R factor is defined as the mean annual sum of individual storm erosion index values, EI_{30} where E is the total storm kinetic energy and I_{30} is the maximum, 30-minute rainfall intensity (Lu and Yu, 2002). A grid of the rainfall erosivity R factor was obtained from the National Land and Water Resources Audit (NLWRA), Australian Natural Resources Data Library (ANRDL) (National Land and Water Resources Audit, 2000). It was generated using the models proposed by Yu and Rosewell that relate monthly storm rainfall erosivity, EI_{30} , to daily rainfall (Lu and Yu, 2002), applied using rainfall estimates interpolated from daily rainfall measurements for the years 1980 to 1999 at a grid resolution of 0.05 degrees

(approximately 5 km) (Jeffrey *et al.*, 2001). The R-factor grid was then re-gridded and supplied at 9 second resolution.

Catchment water balance

The flow regime shapes the physical habitat of streams and consequently plays a major role in the distribution patterns of riverine biota (Poff *et al.*, 1997; Bunn and Arthington, 2002; Naiman *et al.*, 2002). Only major streams and tributaries are regularly gauged so flow regime descriptors must be attributed to stream segments from modelled streamflows. Rainfall-runoff models predict streamflow at daily or hourly time steps, but are not suitable for continental scale application as they normally require model parameters to be calibrated with streamflow records for individual catchments (Croke and Jakeman, 2001). Regionalisation of model parameters (Post and Jakeman, 1996; Hennrich *et al.*, 1999; Post and Jakeman, 1999; Peel *et al.*, 2000) allows runoff to be predicted in ungauged catchments though with considerable uncertainty (Croke and Norton, 2004). Developing regional relationships for catchment response characteristics instead of model parameters reduces uncertainty (Croke and Norton, 2004), but suitable models for continent-wide application were not available. Empirical relationships developed for the Murray-Darling Basin (Croke, 2002) would have required extrapolation well beyond the limits of the data on which they were based.

The amount of runoff available for streamflow is a product of the catchment water balance. Essentially this is the precipitation falling within the catchment less the losses due to evapotranspiration (evaporation and plant transpiration) and changes in soil and groundwater storage (Williamson, 1998). Accordingly, a simple catchment water balance was derived from estimates of monthly runoff computed with the water balance module of the GROWEST program (Nix, 1981; Hutchinson, Nix *et al.*, 2004). This or similar models have been found to reasonably reproduce annual (Atkinson *et al.*, 2002; Stein *et al.*, 2002) or monthly flows (Lewis, 1998; Xiong and Guo, 1999; Jellett, 2005).

GROWEST operates on a weekly time step. It adds rainfall to the previous soil storage, conceptualised as a single “bucket” and removes it by means of evapotranspiration. The soil water surplus or “runoff” is thus that rainfall exceeding “bucket full” after allowing for evapotranspiration. Running the model for the full 52 weeks of the starting year sets the initial soil water storage for model runs. Input data requirements for GROWEST include monthly rainfall and pan evaporation and soil characteristics. Like GROCLIM, GROWEST interpolates monthly input rainfall and evaporation data to a weekly time step but unlike GROCLIM, is not restricted to long-term mean climatic inputs. In order to assess variability in year to year runoff, GROWEST was run for each of the years (1968 to 1998) for which monthly climate surface coefficients (Kesteven *et al.*, 2004) were available. Gridded estimates of monthly rainfall and pan evaporation for the 31 years were derived using the LAPGRD program (Hutchinson,

Table 7.3 Classes of percentage of clay in the A horizon used to assign GROWEST soil texture category. Clay percentage figures were based on values from McKenzie *et al.* (2000).

Clay percentage	GROWEST soil texture category
< 20%	Sandy loam
20 - 40%	Clay loam
≥ 40%	Clay

2004b), the climate surfaces and the DEM, resampled to a spatial resolution of 36 seconds as described above.

GROWEST employs two soil parameters: the maximum available soil water capacity and a parameter that reflects the relative water retention capabilities of the soil. This second parameter determines the shape of the response of the ratio of actual to potential evapotranspiration, as a function of the fractional soil water storage, and is derived from the soil texture category (one of sandy loam, clay loam or clay). Input grids of soil water capacity and soil texture category were derived from the Australian Soil Resource Information System (ASRIS) grids downloaded from the Australian Natural Resources Data Library (ANRDL). Soil texture category was defined by classifying the values in the grid of percent of clay in the A horizon (National Land and Water Resources Audit, 2001f) (Table 7.3). Maximum available soil water parameters were derived by summing the gridded values for the soil A and B horizons (National Land and Water Resources Audit, 2001c, 2001d). ANRDL supplies the ASRIS grids at the relatively coarse spatial resolution of 0.01 degrees of latitude and longitude, matching that of the resampled DEM, but certainly masking considerable sub-grid scale variability in soil properties. However, this is not considered to be a serious concern for the water balance calculations as the model is relatively insensitive to the soil type parameter (Lewis, 1998) and while underestimating actual water capacity, the water capacity values derived from the lookup tables of McKenzie *et al.* (2000), are expected to provide a reasonable lower bound on soil water storage (Ladson *et al.*, 2004).

The catchment water balance of a segment is the sum of the runoff contributions of all segment sub-catchments upstream. Sub-catchment contributions are expressed as a flow volume by multiplying the sub-catchment average grid cell runoff by the sub-catchment area. Sub-catchment contributions were accumulated downstream along the DEM-defined flow pathways and distributed to major anabranches in accordance with stream name (in the ratio of 8 rivers: 4 creeks: 1 unnamed, as discussed in Section 4.2.1.4). The water balance was computed for each month of the 31 years.

Summary measures, describing ecologically significant aspects of the catchment water balance – its magnitude (annual average and monthly maximum) and intra- and inter-year variability – were computed from the monthly accumulated runoff volumes at segment pour-point cells (Table 7.4). The reliability of the estimates derived from the simple catchment water balance

Table 7.4 Catchment water balance attributes used to classify stream segments. Attributes summarise the monthly accumulated runoff volumes computed for each of the years from 1968 to 1998.

Attribute	
Annual mean (GL)	Summer mean (GL)
Monthly mean maximum (GL)	Autumn mean (GL)
Annual variability (Cv)	Winter mean (GL)
Monthly variability (Cv)	Spring mean (GL)
Snow contribution (%)	

model was assessed by comparing them with observed data compiled for the NLWRA extension of unimpaired streamflow project (Peel *et al.*, 2000) (Appendix 2). Most of the summary measures were well estimated by the catchment water balance model. However, none of the commonly used indicators of low flows (e.g. minimum, median) (Grown and Marsh, 2000; Young, Rustonji *et al.*, 2001; Olden and Poff, 2003) were well estimated using monthly rainfall inputs. Accordingly, water balance statistics associated with low flow conditions were not included among the set of classifying attributes (Table 7.4). Variability was described at two time steps by the coefficient of variation (C_v) of the annual and monthly accumulated runoff estimates. Preferred descriptors of variability, such as the S80 (90th percentile-10th percentile/median) or the S50 (interquartile range/median) (Puckridge *et al.*, 1998; Young, 2001) depend on the median and were poorly related to the modelled catchment water balance statistics. The C_v (standard deviation / mean) is also widely used to report flow variability (McMahon, 1979; Grown and Marsh, 2000; Pegg and Pierce, 2002). Because it relies on the mean, the C_v may be skewed by unusually high flow events (Young, 2001) and it behaves poorly for very low mean values, but it was better estimated by the modelled catchment water balance than other measures of flow variability. Seasonal patterns were accounted for by including averages for the four seasons. The benefits of using consistent, easily interpreted statistics continent-wide were judged to outweigh those obtained by using the more usual wet / dry seasonal classification for monsoonal areas of northern Australia.

The distribution of the summary statistic values computed for each of the 1 million stream segments was highly skewed towards smaller values so all volume statistics (+1) were $\log_{10}$ transformed prior to classification to better differentiate smaller streams. A square root transformation was also examined but while this produced a classification that divided larger streams more often, it less reliably recognised desirable partitions between permanent and non-permanent streams.

Precipitation that falls as snow during winter may not produce runoff until it melts in spring. Rapid snowmelt can produce significant flood peaks. The timing of such events in relation to temperature determines their ecological significance (Harris *et al.*, 2000). Ideally, the water balance model would be adjusted directly for these effects, but this would require modelling of snow cover and subsequent snow melt, a major undertaking. Instead, a simple indicator, the proportion of the catchment mean annual runoff generated above the snowline, was devised to distinguish those streams that may be affected by snowmelt. Stream segment values were computed as the percentage of the total accumulated runoff values of upstream grid cells. Long term mean annual runoff was estimated with GROCLIM (Hutchinson, Houlder *et al.*, 2000). The snowline, delineating land usually covered by snow for more than a month each year, is about 1400m on the mainland (Hennessy *et al.*, 2003) and 1100m in Tasmania (Parkinson, 1986). A temperature threshold would provide a more general method of identifying the snowline but appropriate values are difficult to determine. At elevations around the recognised snowline (1350-1450m) in mainland south-eastern Australia, mean annual temperatures vary greatly (2.4 to 10.8 °C). Similarly, long term mean minimum temperature of the coldest month values differ considerably (-0.4 to -5.9°C) although on average they are lower than equivalent values at elevations around the snowline in Tasmania (-3.5 °C and -1.6°C respectively). Only runoff generated in the alpine areas of south-eastern Australia (i.e. areas south of 34°12'S) was considered. Although snow falls occasionally elsewhere along the Eastern Highlands, it has little impact on streamflow characteristics (Warner, 1986).

Geology

The lithological properties of rocks play a major role in shaping river systems, influencing the type of material available for erosion, its weathering and transport (Montgomery, 1999). Stream water chemistry carries the signature of the underlying geology (Smith, 1998; Smart *et al.*, 2001) while ecologically important properties of the stream hydrograph, such as the contribution from groundwater or the size of the peak flows, are related to the type of rocks underlying the catchment (McMahon, 1977; Goodwin, 1996).

Stream segments were attributed with the areal proportion of bedrock lithology classes derived from the digital version of the 1976 Edition of Geology of Australia, 1:2,500,000 scale (Bureau of Rural Sciences, 1991) within both of the upstream catchment and the segment sub-catchment. To better reflect similarities in geochemical or hydrogeological properties among rock types the large number of lithostratigraphic mapping units (> 150) were combined according to the 10 bedrock classes defined by Nix, Stein, Stein and Tassell (1992) (Table 7.5). These classes were formed by hierarchical agglomerative clustering of the map units based on ratings for hardness, grainsize, nutrient supply potential and acidity/ alkalinity. Ratings were computed as the average of the mean, minimum and maximum scores (on a scale of 0.0 to 10.0) allocated to the

Table 7.5 Lithological classes defined by numerical clustering of map units on the digital version of the 1976 Edition of Geology of Australia, 1:2,500,000 scale (after Nix, Stein, Stein *et al.*, 1992)

Class	Description
1	Granitic rocks and high grade metamorphics + minor volcanics
2	Group 1 rocks overlain by calcareous sand
3	Sedimentary rocks + minor volcanics and metamorphics
4	Limestone
5	Acid volcanics and intrusives
6	Intermediate and basic volcanics
7	Basic and ultramafic intrusives
8	Basaltic rocks + minor acid volcanics and intrusives
9	Coastal aeolian calcareous sand and limestone
10	Evaporitic gypsum, halite, clay and sand

individual lithologies that occur within a map unit, weighted according to their relative dominance. While not developed for their significance to river systems, these ratings nonetheless describe relevant characteristics. For example, grain size is related to hydraulic conductivity (or permeability), an important hydrogeological characteristic (Goodwin, 1996). Grain size is also an important determinant of the stream bed particle size distribution, a critical determinant of habitat suitability for many aquatic biota (Ralph Ogden, CSIRO Land and Water pers. comm. 10/10/02).

The clustering of map units by Nix *et al.* (1992) was based on the original digital version of the 1976 edition of the Geology of Australia map (Bureau of Rural Sciences, 1991). This study uses the more recent revision, made available on the web in November 1998. The newer coverage includes 22 map units that were either not assigned a rating by Nix and colleagues or the map unit name and/or description do not match any in the original map. Guided by their lithological descriptions these map units were manually allocated to one of the ten Nix groups.

As the nature of unconsolidated sediments also shapes stream character (Snelder and Biggs, 2002; Jerie *et al.*, 2003), stream segments were attributed with the proportion of their upstream catchment area in each of six surficial cover categories (Table 7.6). Categories were drawn from the surficial cover theme of the digital version of the 1976 Edition of Geology of Australia, 1:2,500,000 scale, released on the web in 1999 (Bureau of Rural Sciences, 1991).

The influence of geological structural controls (e.g folds and faults) on the alignment of streams and the extent of valley dissection (Goodwin, 1996; Alabyan and Chalov, 1998; Brierley and

Table 7.6 Surficial cover categories from the digital version of the 1976 Edition of Geology of Australia, 1:2,500,000 scale (Bureau of Rural Sciences, 1991)

Class	Description
0	Exposed bedrock
1	Quartz, sand
2	Sand, silt, clay and gravel
3	Limestone
4	Ferruginous, aluminous, siliceous duricrust
5	Clay, silt, minor sand

Fryirs, 2002) was not directly attributed to stream segments. Rather it is accounted for both in the characteristics of the derived stream network and topographic attributes.

Terrain

Seven terrain parameters (Table 7.7) were computed to account for the significant influence of topography on stream character and behaviour.

Measures of slope and relief provide an indicator of the energy available for sediment transport and the potential for erosion or deposition (Jerie *et al.*, 2003). They were computed directly from the national 9 second DEM (Section 4.2.1.1). Valley slope has a direct influence on channel slope and hence stream power and velocity (Knighton, 1998). It was calculated by dividing the difference in elevation between the highest and lowest cells of the stream segment by its length. The length of the segment was computed as the sum of the distances to move to the next cell in the direction of flow for all cells in the segment. Very short stream segments of just one cell were assigned the average of the segment slope values computed for the upstream and downstream segments.

The intensity of erosion processes on hillslopes is associated with their slope. Slope also exerts a significant control on hillslope-stream coupling and thus the transfer of material from hillslopes into the channel (Brierley *et al.*, 1996). The average hillslope was computed for each segment sub-catchment. Hillslope grid cells were identified by their Flatness Index class, a terrain index that indicates relative landscape position and the potential for deposition (Appendix 1). Slope was calculated using biquadratic spline interpolation with the SLPGRD program (Michael Hutchinson, CRES pers. comm. 15/10/04). While slopes computed from the 9 second DEM seriously underestimate local slopes, they are nonetheless an important predictor of slope (Gallant, 2001) and can therefore be expected to differentiate the relative strength of erosion processes between sub-catchments.

Table 7.7 Terrain attributes used to classify stream segments.

Attribute	Definition
Stream slope	(Stream segment elevation range / segment length) * 100 (%)
Sideslopes	Segment sub-catchment mean slope for hillslope cells (degrees)
Relief ratio	$R = h / L$ where h = difference in elevation between the highest point on the catchment divide and segment pour-point L = maximum length of the catchment (Gordon <i>et al.</i> , 1992)
Valley confinement	Proportion of stream and immediately neighbouring cells that are not valley bottoms (%)
Catchment storage	Proportion of cells in the catchment that are valley bottoms (%)
Catchment area	Sum of the areas of all grid cells upstream of the segment pour-point (km ²)
Catchment shape (Elongation ratio)	$R_e = D_c / L$ where: D_c = the diameter of a circle with the same area as the catchment area upstream of the segment L = the maximum length of the catchment along a line basically parallel to the main stem (Gordon <i>et al.</i> , 1992)

Measures of relief are indicative of the energy available for sediment transport at broader scales (Jerie *et al.*, 2003). Because runoff is usually quicker in steeper catchments, hydrographs have larger and spikier peaks (Thorne, 2004). Relief was measured by the Relief Ratio, a simple expression introduced by Schumm in 1956 (Gordon *et al.*, 1992). It was computed as the difference in elevation between the highest grid cell on the drainage divide and the elevation at the segment pour-point, divided by the maximum length of the catchment. Maximum length is the longest upstream path length, estimated by summing the distance to move across the surface, allowing for the change in elevation, from the centre of the grid cell to the centre of the next grid cell downstream in the direction of flow.

The size and shape of the catchment influence the water and sediment yield and its timing and distribution. The catchment elongation, for example, has a large bearing on the timing and magnitude of peak flows during flood events (Bárdossy and Schmidt, 2002; Moussa, 2003). A commonly used measure of catchment shape is the Elongation Ratio (R_e) (Gordon *et al.*, 1992; Brierley *et al.*, 1996; Davies *et al.*, 2000; Parsons *et al.*, 2002). It was calculated at each segment pour-point by dividing the diameter of a circle with the same area as the catchment

above the pour-point by the maximum length of the catchment. The more elongated the catchment, the closer the value of this ratio is to zero. Catchment size is described by the total area that drains to the catchment outlet (Gordon *et al.*, 2004). It was computed by accumulating the area of all grid cells upstream of the segment pour-point. The total area upstream of a flow bifurcation was included in the area of all downstream segments (i.e. it was not divided at the bifurcation, see Section 4.2.1.4). Catchment area values were $\log_{10}$ transformed to reduce data skew.

A primary differentiating criteria in many river classification schemes is valley confinement (e.g. Whiting and Bradley, 1993; Rosgen, 1994; Brierley and Fryirs, 2000). The width of the valley relative to that of the channel controls the energy of inundating flows (Fagan and Nanson, 2004) and the potential for channel migration (Pickup and Marks, 2001). Wide floodplains may act as sediment sinks, trapping sediments and sediment-bound nutrients deposited during high flow conditions (Norris *et al.*, 2001) and limiting movement of material from the hillslope into the channel (Pickup and Marks, 2001). Confinement must be described relative to the width of the channel (Jerie *et al.*, 2003), but neither valley nor channel width can be estimated reliably with the available continental scale data. Instead, an indicator of valley confinement was derived by considering the Flatness Index class values (Appendix 1) of the grid cells neighbouring the channel cells. It was computed as the percentage of the cells in the immediate 3 x 3 cell neighbourhood of the channel cell that were not classified as valley bottoms (i.e. cells that were classified as hillslope, ridge top flat or indeterminate). Small values of this indicator thus identify unconfined channels on wide floodplains. Intermediate values suggest the presence of smaller floodplain pockets.

Valley confinement is indicative of the depositional environment at the segment scale. The final terrain parameter, catchment storage, describes the relative proportion of depositional areas in the catchment, calculated simply as the proportion of grid cells in the catchment upstream of the segment pour-point that are valley bottoms.

Vegetation

The type and cover of catchment vegetation have wide ranging influence on river systems. Complex interactions with the fluvial environment exert a major control on the hydrologic and sediment supply regimes and hence the form and behaviour of the river (Brooks, 1994; Brooks and Brierley, 2002). Riparian vegetation supplies organic material to the stream, a source of energy and nutrients to drive riverine food webs (Boulton and Brock, 1999) and woody debris that forms an important habitat component for fish (Koehn and O'Connor, 1990) and many invertebrates (Lake, 1995). The cover and type of vegetation affects microbial diversity and ecosystem functioning (Lecerf *et al.*, 2005), the stability of stream banks (Gordon *et al.*, 2004) and the influence of shading on stream biota (Grouns *et al.*, 2003). The accumulation of woody

debris in the stream may trigger avulsions and abandonment of old channels (Gurnell, 1995) and shapes the development of in-channel features such as pools, bars and steps (Montgomery and Buffington, 1997).

A grid of pre-European vegetation was downloaded from Environment Australia's Environmental Data Directory (<http://www.environment.gov.au/database/edd>). The grid represents the 23 major vegetation groups from the National Vegetation Information System (NVIS) Version 1 (National Land and Water Resources Audit, 2001a) at a grid resolution of 0.01 degrees. Only the dominant vegetation in each grid cell is recorded. Pre-European vegetation, rather than present vegetation, was used to avoid confounding the classification with the effects of recent (post European) industrial society. Stream segments were attributed with the proportion of both the segment sub-catchment and its upstream catchment in each of the 23 vegetation categories.

7.2.2.2. Classification

A large data matrix was assembled as input to the ALOB clustering module of PATN (Belbin, 1993b). The rows in the data matrix represent individual stream segments while the columns contain values for the variables from each of the five attribute groups above (climate, catchment water balance, geology, terrain and vegetation).

ALOB employs a simple iterative procedure to allocate objects into groups (Box 7.2). As it is relatively insensitive to the choice of initial seeds (Belbin, 1987), it was run with the default option of simply selecting the first object in the data matrix as the initial group seed. Objects that are more distant than the specified allocation radius from this initial seed become the seeds for additional groups. Distance is estimated in data space according to the specified association measure. The number of groups created depends on the value of the allocation radius: the larger the radius the fewer groups are seeded. Alternatively, ALOB can automatically determine a radius to produce the desired number of groups. Stopping rules are invoked either by setting a fixed number of iterations or a minimum number of reallocations. In all cases the number of iterations was set to a number large enough to ensure that the iterative reallocation procedure would continue until complete, that is, until either there were no more objects to reallocate or oscillations (objects transferring back and forth between two groups) were detected.

The association between an individual object and the group was computed with the Gower metric (Gower, 1971). This measure is well suited to quantitative data (Belbin, 1993b) and incorporates a range standardisation ensuring attributes with a larger range of values do not have undue influence on the classification. It does not account for data skewness, but as described above, this was removed by transforming selected attributes. Thus the distance D_{ij} between objects i and j for p variables x_k ($k=1, \dots, p$) is given by

Box 7.2 The ALOB clustering algorithm (Belbin, 1987).

1. Define a set of seed objects to act as a starting configuration
2. Create additional seeds from objects that are further (in data space) than the allocation radius from existing seeds
3. Generate an initial classification by allocating each object to the closest seed
4. Compute the centroid of each of the groups as the average of the attribute values for all objects in the group
5. Sequentially extract each object from its group and re-compute the group centroid. Calculate the distance from the object to each group centroid and (re-)allocate the object to its closest group
6. Recompute all group centroids and repeat steps 5 and 6

$$D_{ij} = \sum_{k=1}^p \frac{|X_{ik} - X_{jk}|}{r_k}$$

Where r_k is the range of the k th variable (Belbin, 1993b).

Two alternative approaches to develop a classification of stream segments were tested (Figure 7.1). Firstly, a single classification (ALL) was produced with segments grouped according to their association computed across all selected attributes (Table 7.8). Attributes were weighted to ensure attribute groups contributed equally to the measure of dissimilarity regardless of the number of attributes in the group. This strategy is analogous to the environmental domain approach of Mackey and colleagues (Mackey *et al.*, 1988; Mackey *et al.*, 1989) and the global agroclimatic classification of Hutchinson *et al.* (1992).

The IBRA classification (Environment Australia, 2000) underpins the development of the National Reserve System program in Australia. To facilitate comparison with this terrestrially focussed classification, the number of groups for the combined classification (ALL) was set to the number of IBRA sub-regions (354). This number was too small, however, to adequately represent the spatial variation across all attributes. The resulting groups were very heterogenous especially for categorical variables. Accordingly, the proportion of the catchment within each vegetation type was omitted to reduce the number of attributes on which the combined classification was based.

Table 7.8 Attribute sets used to generate each classification. Darker shaded cells indicate the attribute group for the corresponding classification. Attributes within a group are described in Section 7.2.2.1

Attribute group	Classification					
	ALL	Climate	“Flow”	Geology	Terrain	Vegetation
Climate						
Catchment water balance						
Geology	1			2		
Terrain						
Vegetation						2

1. Attributed as proportion of upstream catchment area in class
2. Attributed as dominant class in segment sub-catchment

The second classification strategy (Figure 7.1) produces five classifications of 10 groups, one for each of the major attribute groups (Table 7.8). These classifications combine hierarchically, reflecting the relative scale of their influence (*sensu* Snelder and Biggs, 2002) on stream ecosystem patterns and processes. Hence, one or more factors may classify a stream segment, the classification being expressed as the combination of its group membership at the selected levels. The classifications for three of the levels, climate, flow and terrain, were derived with the ALOB non-hierarchical clustering algorithm with all attributes in the respective attribute group weighted equally. Representation of all combinations of the substrate categorical variables (5 surficial geology in combination with 10 lithology categories) is not possible in just 10 groups. Instead, the geology classification of stream segments derives simply from the dominant lithological class in the segment sub-catchment (i.e. the class occupying the largest area). Similarly, the dominant vegetation category in the segment sub-catchment defines the vegetation class. The original NVIS groups were first combined, primarily according to their structural type, to form 10 categories (Table 7.9). Group number 10 (originally NVIS group 23) is a mixed group including mangroves and many non-vegetated categories. This is not expected to create any difficulties for classification, however, because other factors, such as climate or topography, clearly differentiate streams within this vegetation group.

The number of groups was set at 10 because this is the largest number that can be represented by a single digit (0-9) and hence produce a combined code that is readily interpreted and queried. A larger number of groups would likely produce too many combinations to be reasonably tested – even at 10 groups there are 10^5 possible combinations for a five level hierarchical classification. A 10 group classification is also easily described and communicated.

Table 7.2 The ten vegetation groups formed from the combination of the National Vegetation Information System (NVIS) major vegetation classes.

Group	Description	NVIS classes
1	Rainforest and vine thickets	1
2	Eucalypt tall open forests and open forests	2, 3
3	Eucalypt low open forests, woodlands and open woodlands, Tropical eucalypt woodlands/grasslands, Mallee woodlands and shrublands	4, 5, 11,12,14
4	Non-eucalypt forests and woodlands (Acacia, Callitris, Casuarina)	6,7,8,9,10,13
5	Low closed forests and closed shrublands	15
6	Acacia and other shrublands, heath	16, 17, 18
7	Tussock and hummock grasslands	19, 20
8	Other grasslands, herblands, sedgelands and rushlands	21
9	Chenopod shrubs, samphire shrubs and forblands	22
10	Mangroves, tidal mudflats, samphires and bare areas, claypan, sand, rock, salt lakes, lagoons, lakes	23

Of course, 10 groups are arbitrary and may not produce the most ecologically appropriate partitioning. To explore whether a larger number of groups might produce a better partitioning of stream segments, 12 classifications of between 5 and 60 groups were generated with ALOB for each of the attribute sets with continuous variables (i.e. climate, terrain, catchment water balance). Where no value of the radius will produce exactly the desired number of groups ALOB will use that value which generates the closest number of groups. Hence, the numbers of groups tested only approximately increment by five. Due to the very long processing times involved (more than a week for a single classification of 355 groups) alternative numbers of groups for the combined classification (ALL) were not tested.

7.2.3. Evaluating the classifications

7.2.3.1. Characterising the groups

Evaluation of the classifications was directed at interpretation - characterising the groups and identifying the principal attributes that distinguish them. Box and whisker plots were produced using the R statistical software package (R Development Core Team, 2004). Kruskal-Wallis rank sum tests were used to test the null hypothesis that the distribution of the attribute values is the same in each group (R Development Core Team, 2004). Kruskal-Wallis is a non-parametric

statistic that is more robust to departures from normality than the F-statistic (Belbin and Collins, 2004). The group means of the most similar groups identified from the dendrogram (see below) were also compared to ensure differences were ecologically meaningful and greater than the measurement error of the variable.

To assist understanding of the relationships between the groups, the centroids of the groups generated with the ALOB non-hierarchical algorithm were themselves classified using the hierarchical agglomerative clustering routine, flexible UPGMA (Un-weighted Pair Group Mean Averaging), implemented in the FUSE module of PATN (Belbin *et al.*, 1992). This clustering strategy weights objects equally as groups are successively combined. Belbin and colleagues (1992) demonstrated that this approach recovered simulated clusters better than an alternative option, flexible WPGMA, that weights groups equally, implicitly varying the weight of the object with the size of the group. Input to FUSE was the association matrix containing the dissimilarity between all pairs of group centroids produced by ALOB using the Gower metric and the specified attribute groupings. The β parameter, controlling the degree of dilation or contraction of the distance between groups as they join (“fuse”), was set to the recommended value of -0.1 (Belbin *et al.*, 1992). A dendrogram depicts the fusion history graphically.

A three dimensional ordination of the groups was produced from the inter-group association matrix using the semi-strong hybrid multidimensional scaling routine (SSH) in PATN (Belbin, 1991) with the default options. The ordination enables further exploration of the group structure and provides a mechanism for automatic assignment of mapping colours that reflect the affinity between groups. Correlation of the attributes with the ordination axes was computed using the PCC routine in PATN. As this analysis was exploratory, tests of significance were not conducted. Plots of the ordination were produced using the enhanced graphical display capabilities of the new version of PATN, released for the Windows operating environment in 2004 (Belbin and Collins, 2004).

The classification was mapped in ArcGIS (ESRI, 2005) by associating the group affinity vector, identifying the group allocation of each stream segment, with the value attribute table of the stream segment grid. Mapping colours were assigned by scaling the red, green and blue colour components according to the group’s scores on the three ordination axes (Belbin *et al.*, 1983).

7.2.3.2. Numbers of groups

To address the question of whether 10 groups adequately partitions the variation in landscape attributes among stream segments, two criteria were assessed: the average within group distance of an object from its group centroid (“average deviation”) and a modified measure of classification strength.

The average deviation is a measure of the heterogeneity of a group; larger values signify more heterogeneous groups. ALOB computes average deviation values using the selected dissimilarity measure, in this case the Gower metric. For each classification the average deviation value at the end of the last iteration was extracted from the ALOB log file and plotted against the number of groups. Average deviation decreases with increasing numbers of groups. A break in the slope of this relationship indicates the level of partitioning at which increasingly finer partitioning would produce relatively smaller reductions in within group heterogeneity. The marginal improvements in within group homogeneity with further sub-divisions may not justify the increased complexity of the classification.

The desire to produce more homogenous groups needs to be balanced against the competing requirement to maintain recognisable differences in landscape characteristics between groups. This trade off was explored by examining plots of classification strength (CS) (Van Sickle, 1997; Van Sickle and Hughes, 2000) against the number of groups. CS measures within group homogeneity relative to the distance between groups. It is normally calculated from the mean of pairwise within group similarities ($\bar{W}$) relative to the mean of all between group similarities ($\bar{B}$) (Van Sickle, 1997; Van Sickle and Hughes, 2000). As this is impractical with very large numbers of objects, the object to group tabulation function in ALOB was tailored to calculate CS using object to group centroids. Mean within group similarity ($\bar{W}$) was computed as the mean similarity of segments from their group centroid and mean between group similarity ($\bar{B}$) as the mean similarity of a segment to all other group centroids. Similarity was calculated as one minus the Gower metric dissimilarity value.

7.3. Results

7.3.1. River Environment Types

A large data matrix comprising 955,632 rows (stream segments) by 43 attributes was assembled for numerical clustering. There are similarities in the spatial distribution of values among attributes within a group; temperature attributes, for example, reflect the strong influence of latitude (Figure 7.2). Nevertheless, the maps display sufficient differentiation between attributes to justify their separate inclusion (Figure 7.2 to Figure 7.4).

Six separate classifications of the stream segments were produced: five classifications of 10 groups, one for each set of attributes; climate, modelled catchment water balance (referred to hereafter as “flow”), geology, terrain and vegetation that together will form the five level hierarchical classification (Section 7.3.2) and a sixth based on all attributes combined (“ALL”) that produced 355 groups. Figure 7.5 to Figure 7.9 map the individual classifications of stream

segments that form the hierarchical landscape classification. Characteristics of the groups in the climate, flow and terrain classifications are summarised in Table 7.10 to Table 7.12.

Descriptions of the geology (Table 7.5) and vegetation classes (Table 7.9) were presented earlier and in more detail in the original documentation (Nix, Stein, Stein *et al.*, 1992; National Land and Water Resources Audit, 2001a). For the sake of brevity, the 355 groups of the ALL classification are not individually described. However, the spatial distribution of the major groupings is apparent when groups are mapped with colours that reflect their affinities (Figure 7.10).

Table 7.3 The climate classification groups.

Group	No. of segments	Description
1	126,595	Hot for most of the year, seasonally wet/dry, high radiation, moderate potential for megatherm plant growth
2	203,923	Hot, seasonally cool, dry, high radiation, moisture limits plant growth
3	20,340	Warm to hot and wet to very wet, seasonally dry or drier, rainfall often high intensity; highest potential for megatherm plant growth
4	95,938	Hot all year, wet, pronounced seasonal wet/dry regime, rainfall has high erosive potential
5	79,332	Warm to hot, moderate rainfall seasonally dry or drier, mixed growth potential mesotherm and megatherm plants
6	200,819	Warm to hot, very dry, low plant growth potential, colder minimum temperatures than group 2
7	28,625	Warm, reasonably wet throughout the year, highest growth potential for mesotherm plants
8	91,173	Warm, cooler minimums and drier than group 7
9	89,213	Cool, no seasonal dry period, good growth potential for microtherm plants
10	19,674	Cold, wet all year, lowest radiation, highest growth potential for microtherm plants

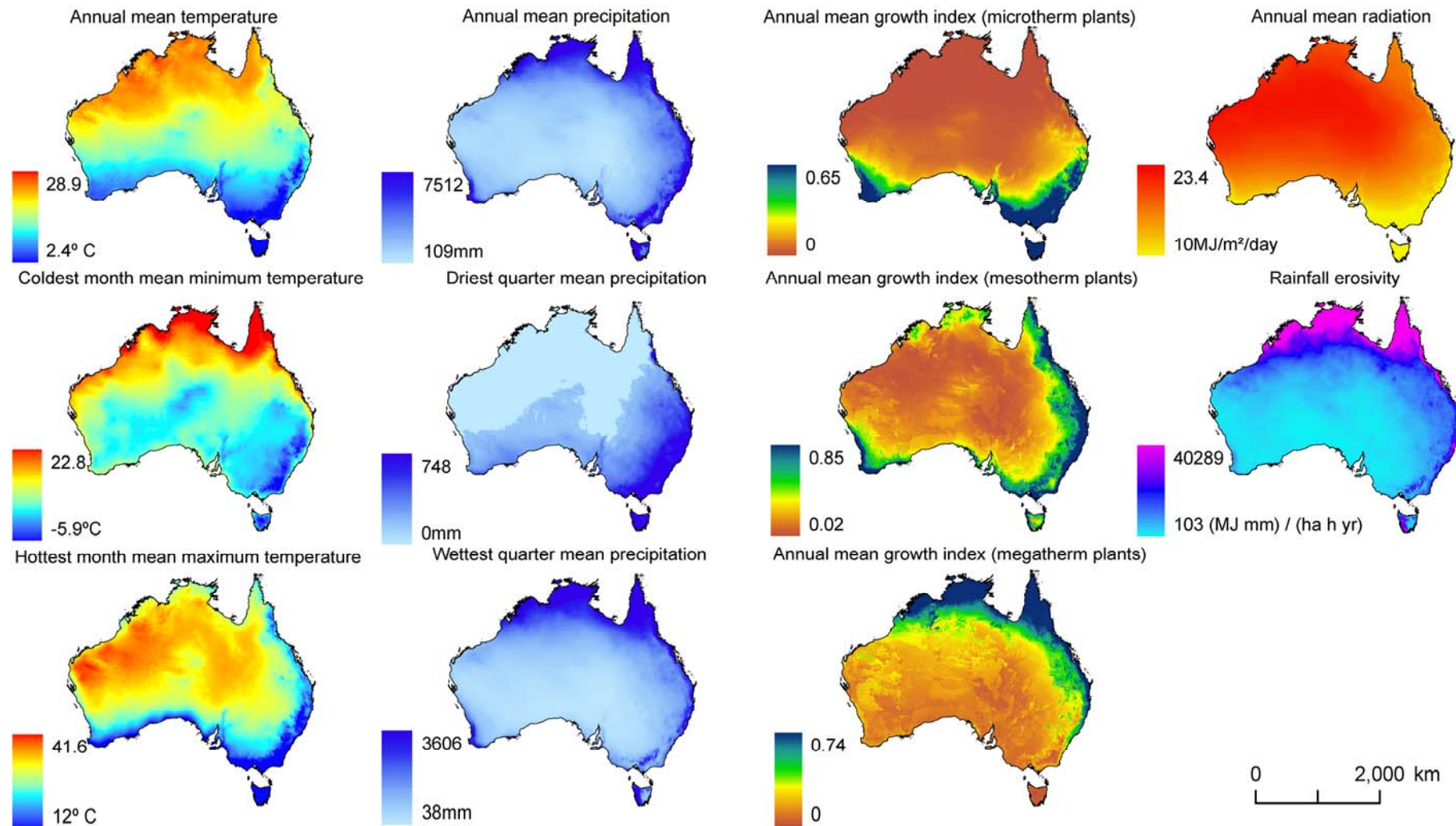


Figure 7.2 Climatic attributes used to classify stream segments. To highlight spatial variation, the grid cell values are mapped using a continuous colour ramp and a linear stretch of values between the mean plus or minus two standard deviations. Stream segments were characterised by the average value of grid cells comprising the segment or its sub-catchment (Table 7.2).

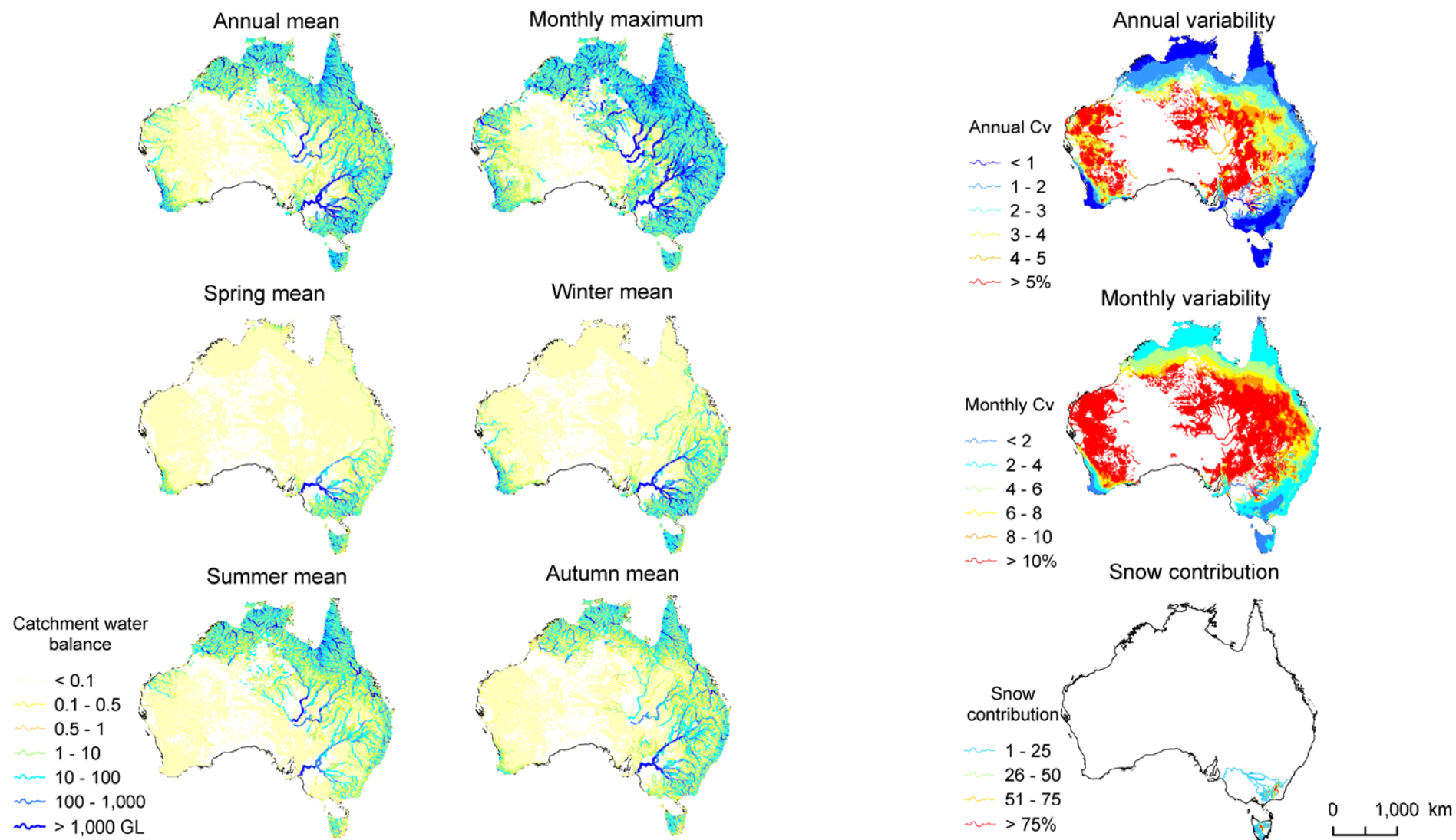


Figure 7.2 Stream segment pour-point values of the “flow” attributes derived from the modelled catchment water balance and used to classify stream segments

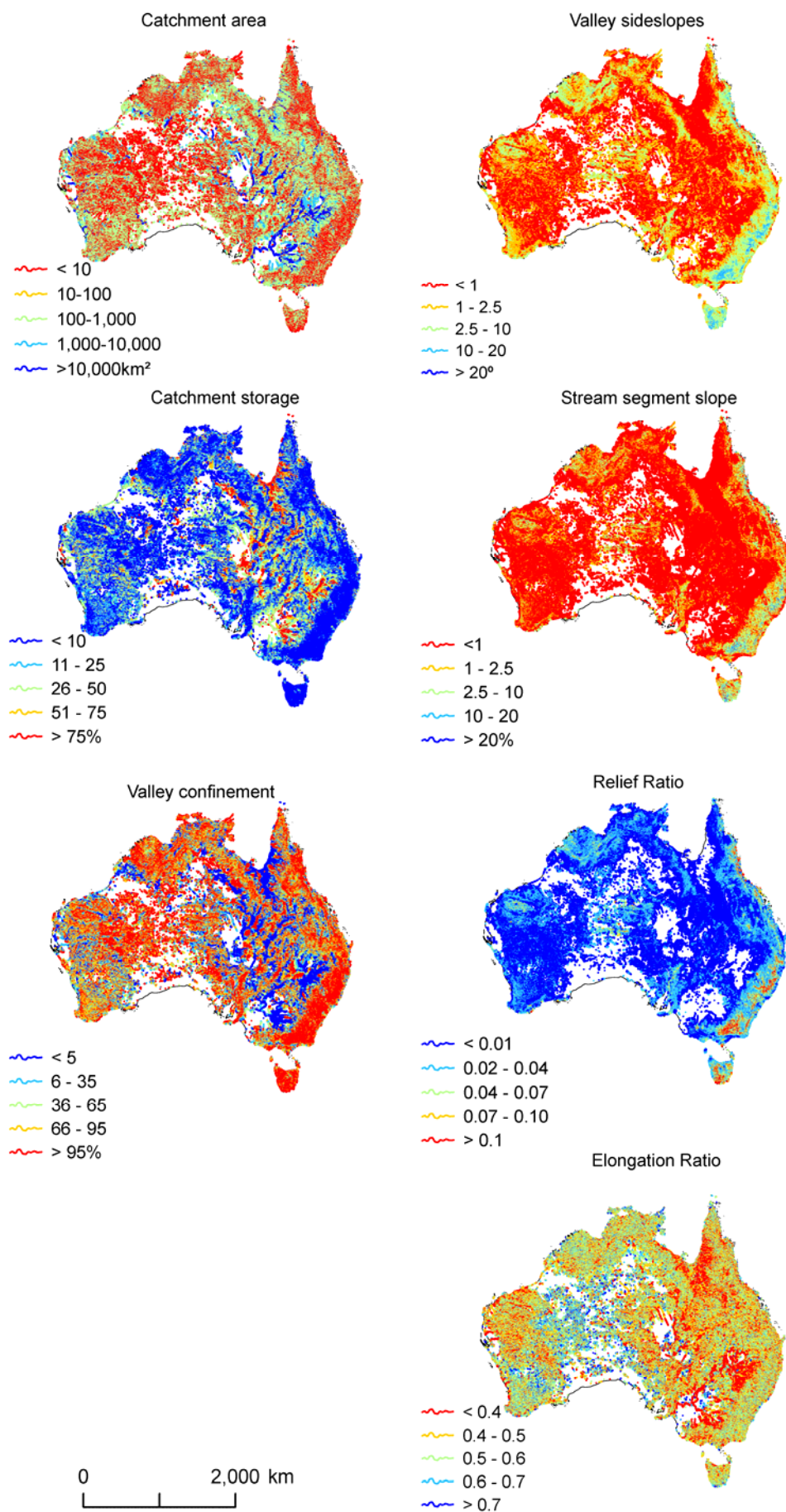


Figure 7.3 Stream segment values of the terrain attributes used to classify stream segments

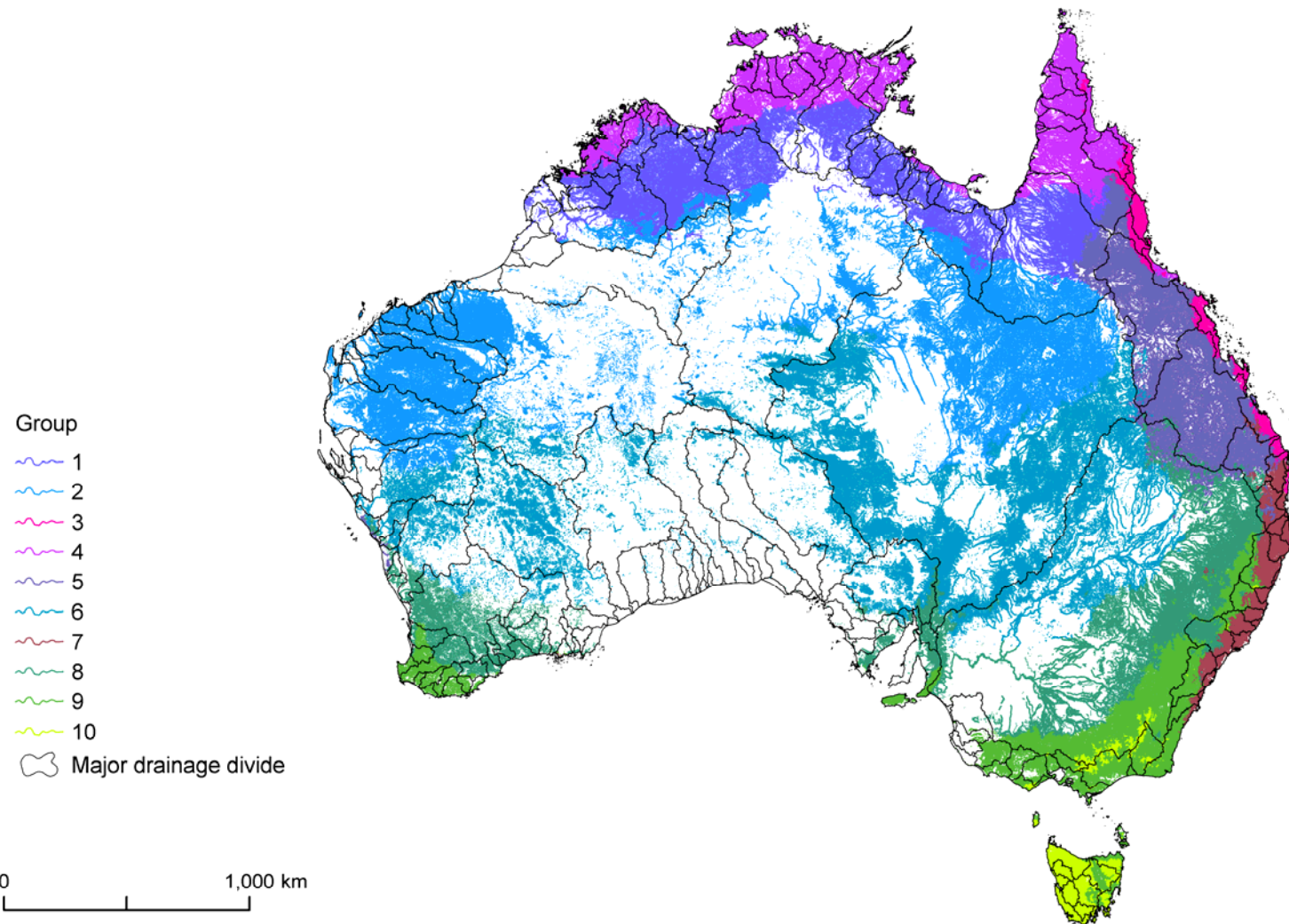


Figure 7.4 The climate classification: stream segments classified by climatic attributes

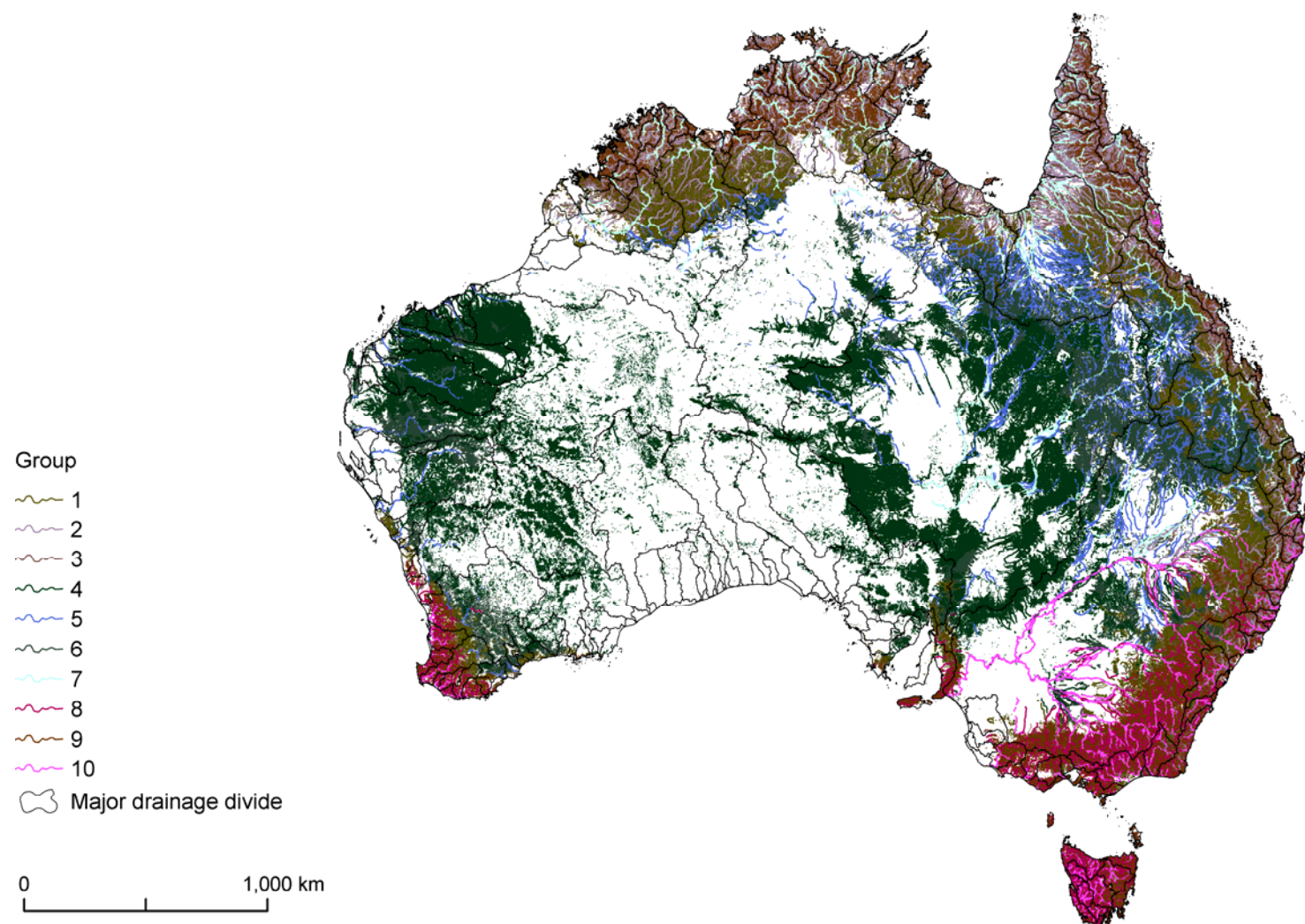


Figure 7.5 The flow classification: stream segments classified by modelled catchment water balance statistics.

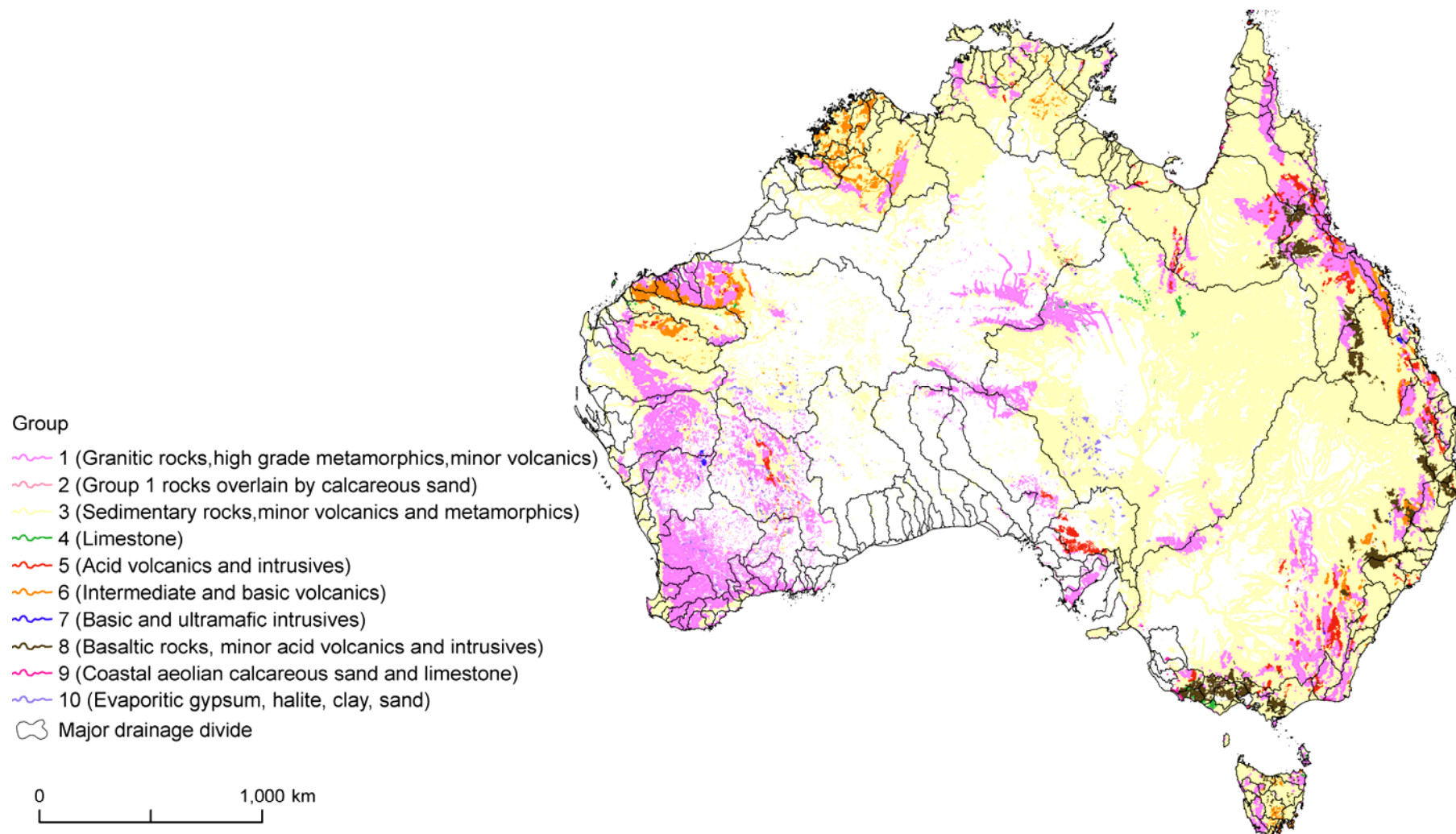


Figure 7.6 The geology classification: stream segments classified by dominant lithological group in the segment sub-catchment

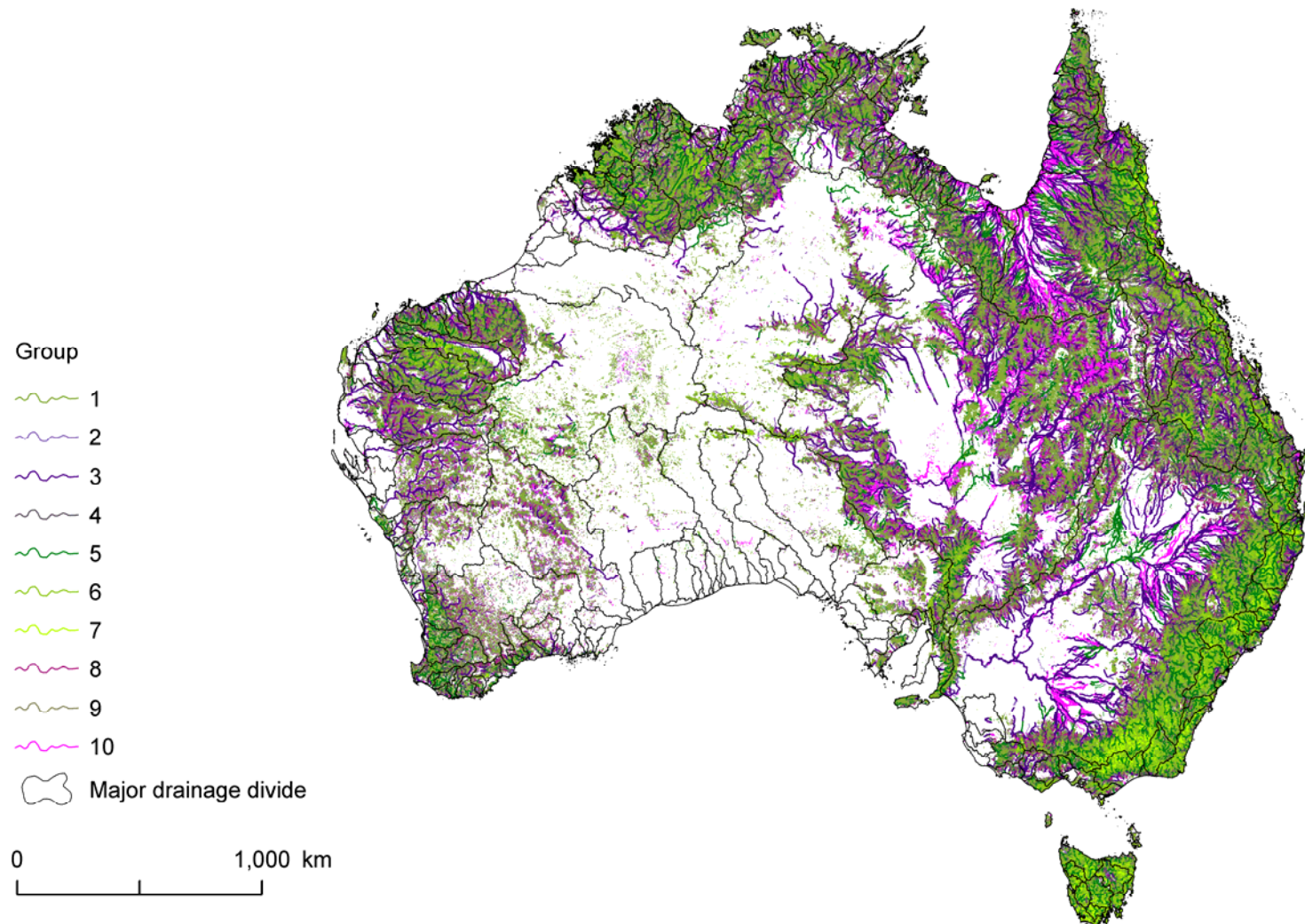


Figure 7.7 The terrain classification: stream segments classified by terrain attributes

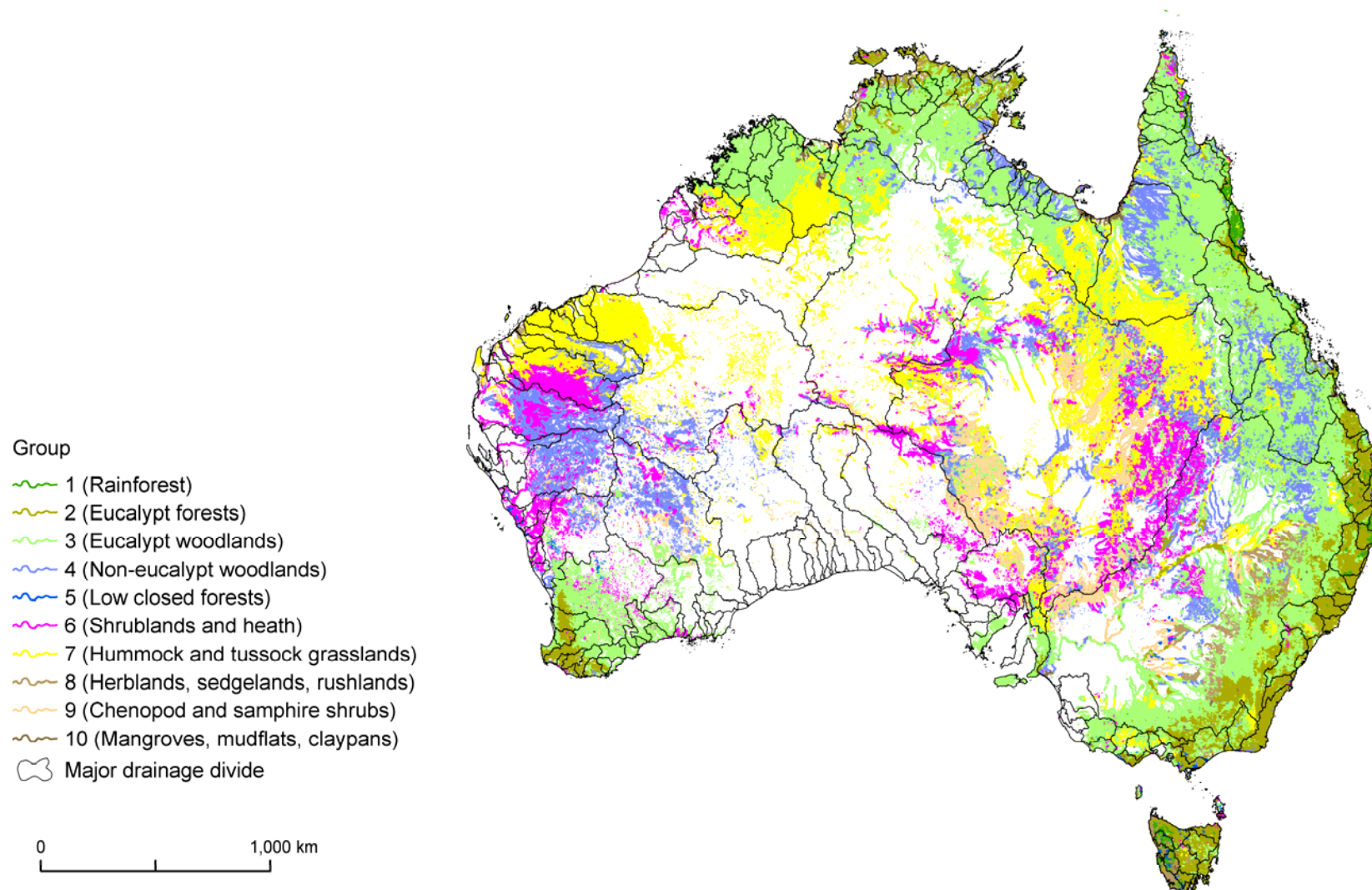


Figure 7.8 The vegetation classification: stream segments classified by dominant vegetation group in the segment sub-catchment

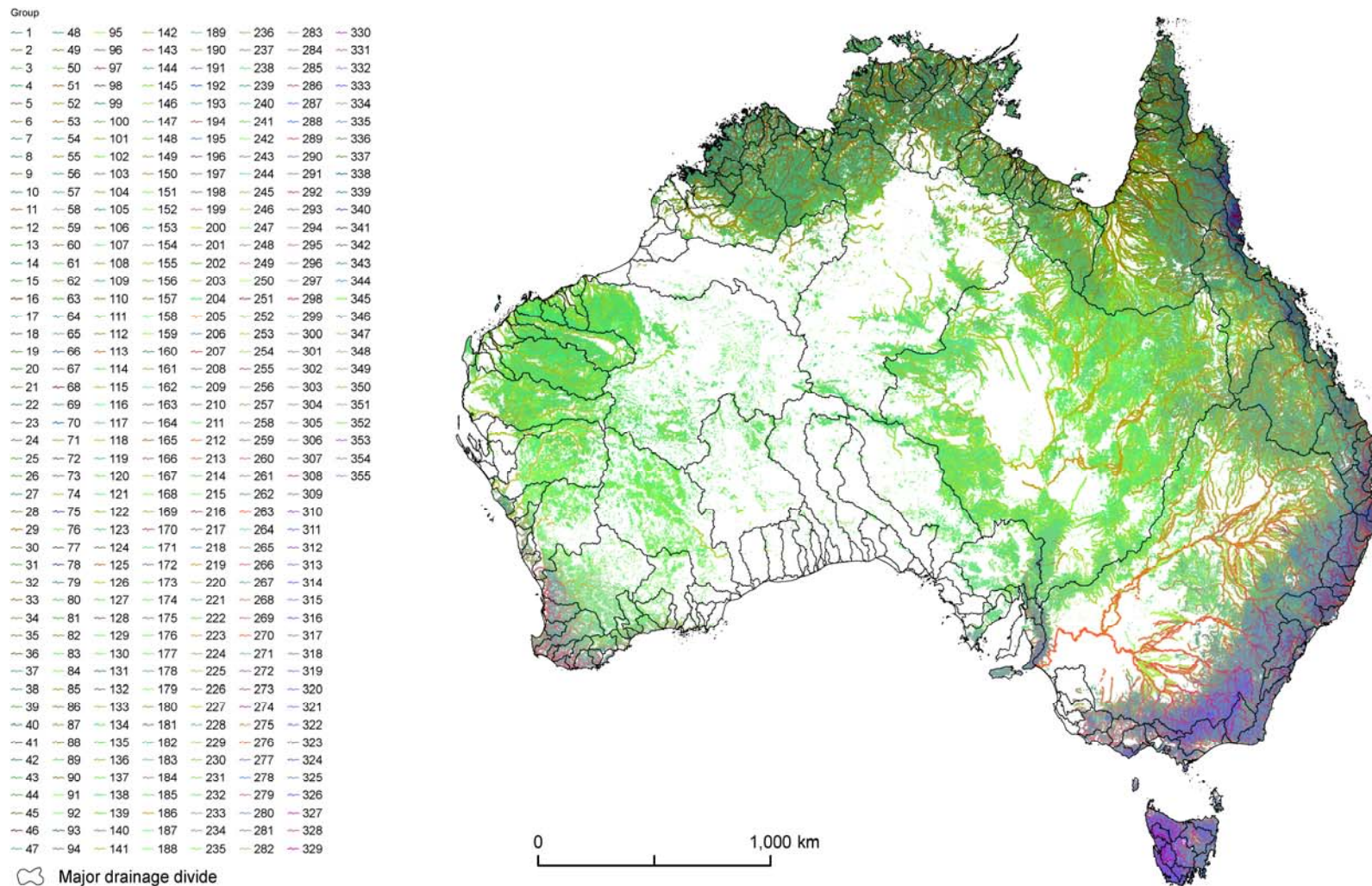


Figure 7.9 The ALL classification: stream segments classified by climate, flow, terrain and geology attributes.

Table 7.4 The flow classification groups.

Group	No. of segments	Description
1	172,482	Small flows, summer dominance
2	33,056	Moderate flows, summer and autumn dominance
3	71,221	Small to moderate flows, summer to autumn dominance
4	314,824	Very low mean flows, no seasonality, extremely variable
5	24,268	Small to moderate flows, higher peak flows, summer dominance, high to very high variability
6	132,680	Low mean flows, summer and autumn dominant, larger peak flows, high to very high year to year variability
7	13,051	Largest mean and peak flows, summer to autumn dominant
8	20,079	Moderate flows, lower in summer, low variability, smaller magnitude peak flows relative to mean
9	165,502	Small flows, reasonably uniformly throughout the year, low variability year to year
10	8,469	Large flows most of the year, lowest in summer, snow melt more likely to influence runoff patterns

A single river may traverse many groups. The Shoalhaven River in south-eastern New South Wales for example, flows through a cool, wet climatic zone (climate group 9) for much of its length, before it enters the southern limits of the warmer, sub-tropical group number 7. Its source is in the coldest climatic zone (climate group 10). As the river flows downstream it moves through four of the ten flow groups, reflecting the increasing discharge with tributary inputs and it has segments that are members of seven of the terrain groups. The Shoalhaven River traverses 12 of the 355 groups in the ALL classification, and more if its tributaries are also considered.

Table 7.5 The terrain classification groups.

Group	No. of segments	Description
1	336,327	Confined valley settings of low to moderate slope; small catchments with little depositional area and low to moderate relief
2	42,226	Partly confined valley settings of low slope; generally small catchments with major depositional areas and low relief
3	45,620	Unconfined valley settings of very low slope; very large, elongated catchments with generally low relief and moderate to large depositional areas
4	73,920	Partly confined valley settings; medium to large catchments of low to moderate relief
5	56,095	Confined or partly confined valley settings of gentle slope with steeper hillslopes; large, elongated catchments of generally low relief
6	128,231	Steeper confined valleys with steep to very steep hillslopes; small catchments of moderate to high relief and very minor depositional areas
7	29,127	Very steep, confined valleys with very steep hillslopes; small catchments of high relief and negligible depositional area
8	92,774	Generally unconfined valleys of gentle slope; medium sized catchments with little relief and moderately large depositional areas
9	109,321	Mostly confined valley settings typically with floodplain pockets; usually small catchments of gentle relief
10	41,991	Unconfined valley settings with little slope; small catchments predominantly in depositional environments

Box and whisker plots are presented for the 10 group classifications generated by direct numerical clustering of the stream segments: climate (Figure 7.11), flow (Figure 7.12) and terrain (Figure 7.13) and for selected attributes for the ALL classification (Figure 7.14). The boxes indicate the inter-quartile range with the central bar signifying the median value. Whiskers are drawn to include all data points that are not more than 1.5 times the inter-quartile range from the box. Outliers are portrayed as separate points in red.

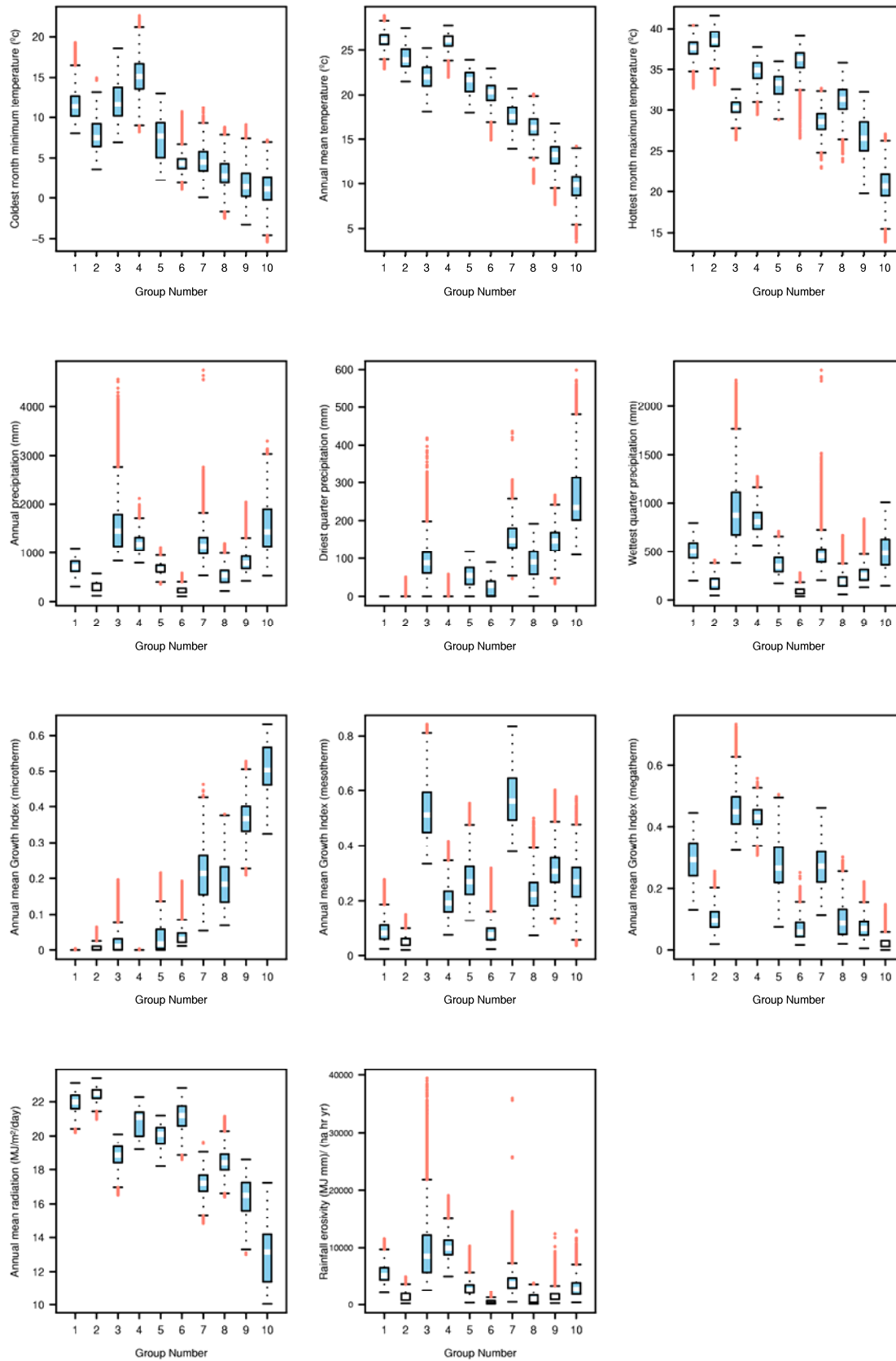


Figure 7.10 Boxplots showing the distribution of climate attribute values among the 10 groups of the climate classification.

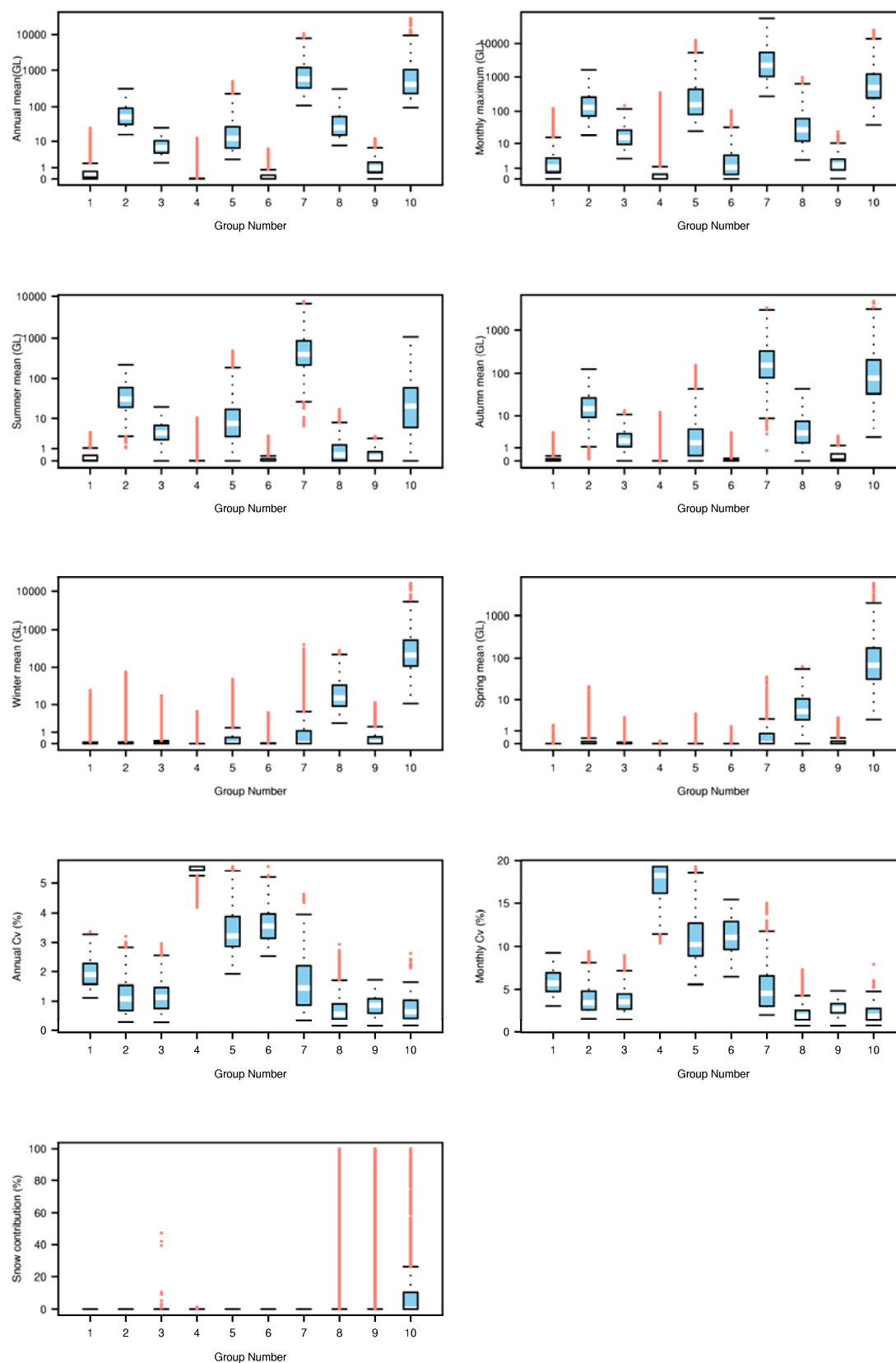


Figure 7.11 Boxplots showing the distribution of catchment water balance attribute values among the 10 groups of the flow classification.

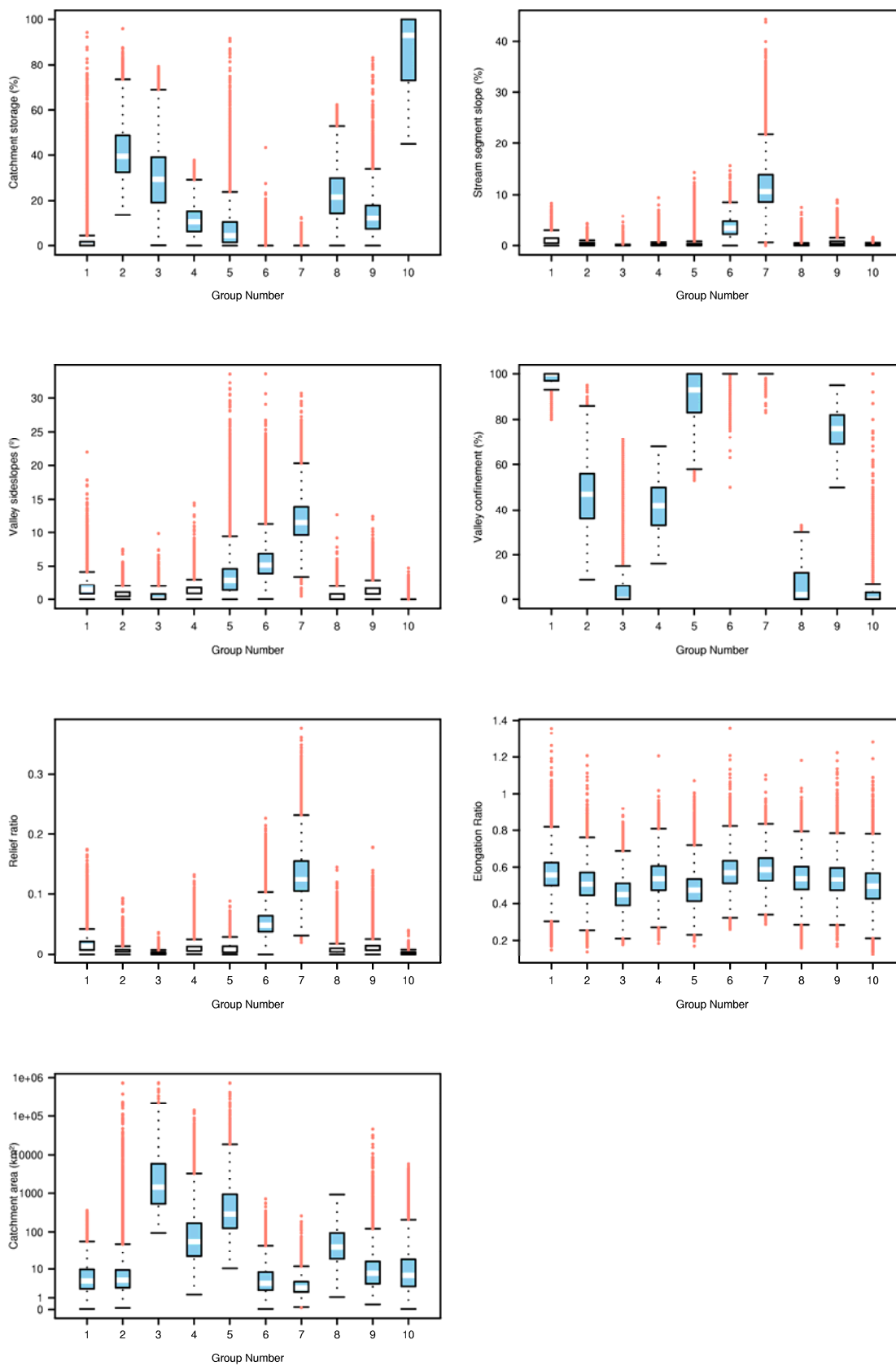


Figure 7.12 Boxplots showing the distribution of terrain attribute values among the 10 groups of the terrain classification.

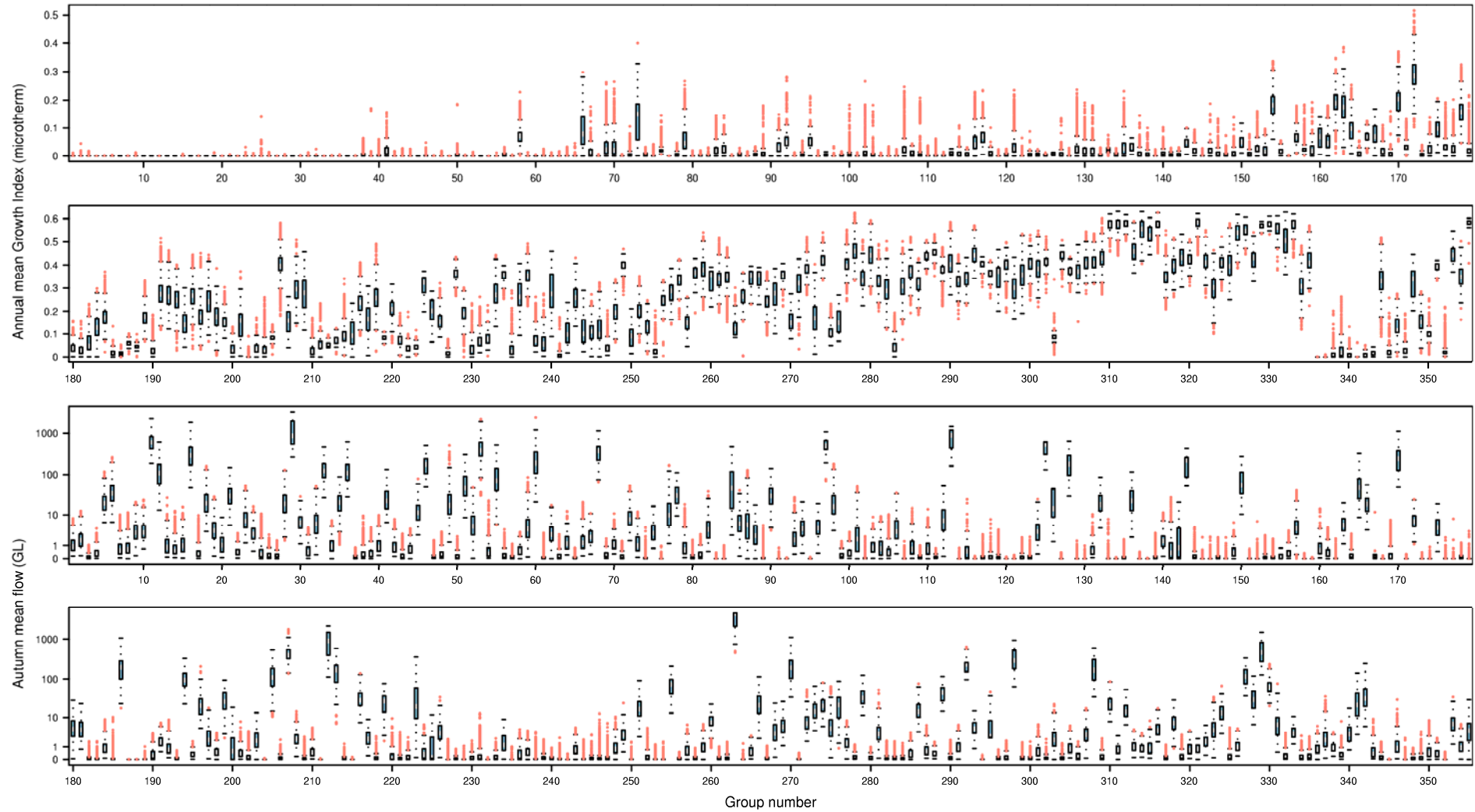


Figure 7.13 Boxplots showing the distribution of values of the climate and flow attributes with the highest Kruskal-Wallis statistic among the groups of the ALL classification

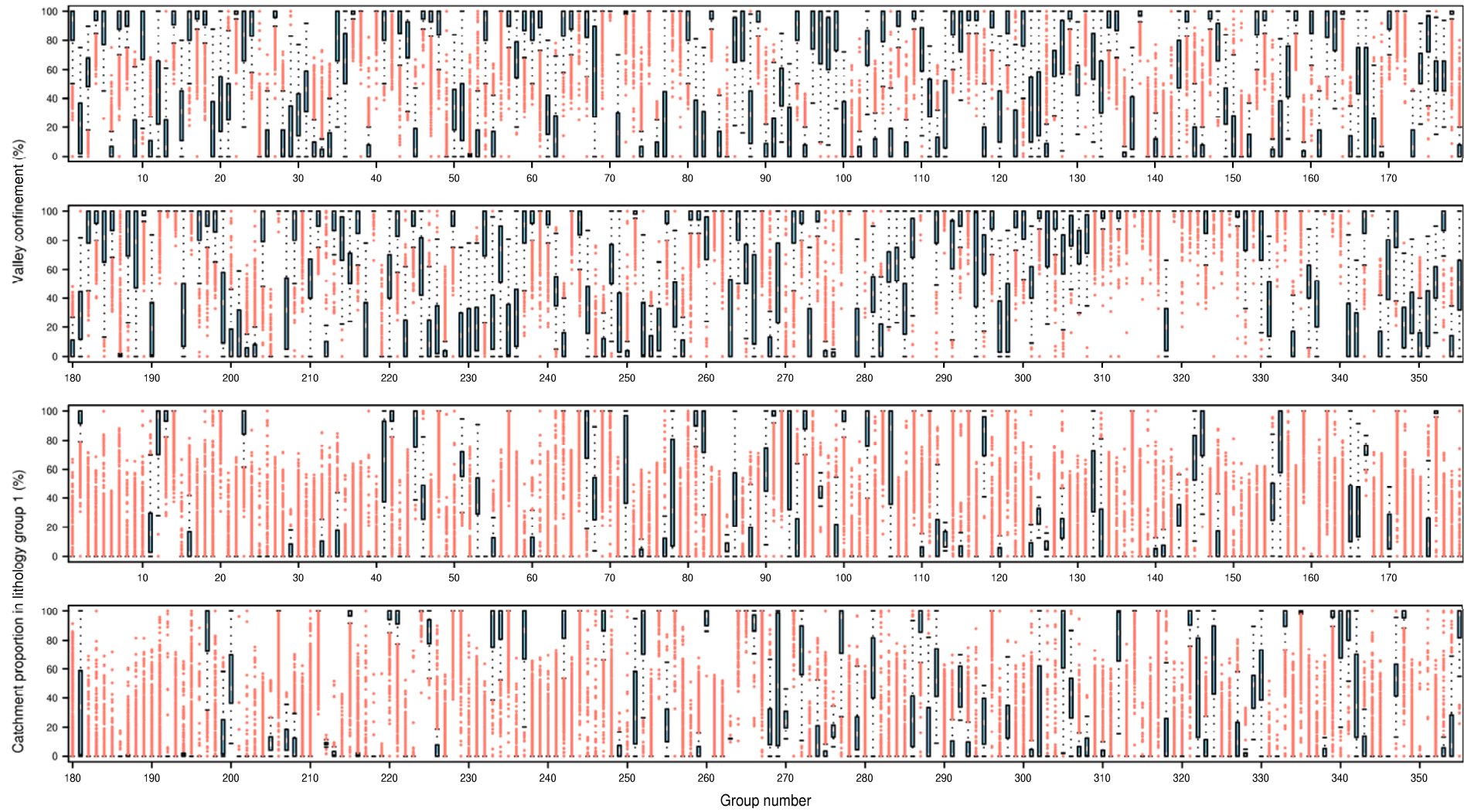


Figure 7.14 (cont.) Boxplots showing the distribution of values of the terrain and geology attributes with the highest Kruskal-Wallis statistic among the groups of the ALL classification

The boxplots show that the groups are clearly differentiated by the chosen attributes. Across most climatic attributes there is little overlap in values between climate groups although at least three groups (numbers 3, 7 and 10) include outliers on the rainfall attributes (Figure 7.11). Groups are also well separated by the descriptors of the flow regime in the flow classification (Figure 7.12) even though there is greater overlap on some attributes. Several groups for instance have low winter and spring mean “flows”. The groups in the terrain classification are generally less compact, all with outliers on one or more of the terrain attributes (Figure 7.13). In particular, every terrain group exhibits a wide range of values of catchment shape as measured by the elongation ratio.

Although a much finer sub-division of the stream segments, the 355 ALL groups also contain a number of outliers. Box and whisker plots are presented for the attribute with the highest Kruskal-Wallis value in each attribute set in Figure 7.14. Climatic attributes produce the six largest Kruskal-Wallis values (Table 7.13), the rankings providing an indication of relative significance in differentiating groups (Belbin and Collins, 2004). Consistent with the trends across the ten group classifications the ALL groups are more clearly differentiated on climatic and flow attributes than on terrain attributes (Figure 7.14). It is difficult to ascribe a single dominant lithological or surficial cover class to each group. The box plots show that the stream segments within a group come from catchments of a variety of substrate types.

Even so, the results of the Kruskal-Wallis rank sum tests (Table 7.13) confirmed for all classifications that the distribution of attribute values was significantly different in at least one of the groups derived by numerical clustering. Although the most similar groups overlap on many attributes the means for at least one are different enough to be ecologically meaningful and greater than the uncertainty associated with its estimation (Table 7.14). The two closest terrain groups (1 and 6) for example, differ only in their stream segment and valley sideslope values but these differences (2.6% and 4.2° respectively) are likely to be enough to significantly differentiate stream power and hillslope erosion processes. The combined classification (ALL) partitions stream segments more finely and as expected, groups are generally closer in environmental characteristics. Group mean values for the closest groups, numbers 22 and 27, are similar for one of the temperature attributes but differ by more than 1.4 to 2°C on the other two attributes (Table 7.14). These differences are well outside the 0.5°C error on the temperature surface estimates (Hutchinson, Houlder *et al.*, 2000).

Table 7.6 Kruskal-Wallis rank sum statistics. The probability value of the test statistic was less than 0.001 in all cases. Attributes are listed in rank order within attribute groups according to the Kruskal-Wallis statistic for the corresponding 10 group classification.

Attribute	Classification			
	All	Climate	Flow	Terrain
Annual mean temperature	859140	867239		
Growth Index (microtherm plants)	882223	861383		
Hottest month mean maximum temperature	805134	819849		
Annual mean precipitation	870885	815167		
Wettest quarter mean precipitation	861538	799144		
Coldest month mean minimum temperature	838699	793170		
Annual mean radiation	796205	791656		
Driest quarter mean precipitation	805279	786621		
Growth Index (mesotherm plants)	802242	770294		
Rainfall erosivity	838784	756493		
Growth Index (megatherm plants)	805175	744271		
Autumn mean “flow”	824078		763532	
Annual mean “flow”	813191		757406	
C _v Monthly “flow”	767284		734744	
C _v Annual “flow”	761243		733899	
Maximum monthly “flow”	689864		573033	
Summer mean “flow”	779369		546577	
Spring mean “flow”	771833		414487	
Winter mean “flow”	797474		271343	
Snow contribution	593340		215549	
Valley confinement	734410			842687
Catchment storage	656674			769119
Segment slope	592431			592956
Relief Ratio	616245			543880
Catchment area	532038			499003
Hillslope mean slope	559861			539405
Elongation ratio	116817			98935
Lithology group 1	781353			
Lithology group 3	780704			
Surficial geology group 0	780597			
Surficial geology group 4	663939			
Surficial geology group 1	663602			
Lithology group 8	597530			
Surficial geology group 2	526025			
Lithology group 6	506112			
Lithology group 5	460660			
Surficial geology group 5	436060			
Lithology group 10	307748			
Lithology group 4	212794			
Lithology group 7	175955			
Surficial geology group 3	80434			
Lithology group 9	55121			
Lithology group 2	24909			

Table 7.7 Group means for closest groups. See Section 7.2.2.1 for a description of attributes and measurement units.

Attribute	Closest groups							
	ALL		Climate		Flow		Terrain	
	22	27	2	6	1	9	1	6
Coldest month minimum temperature	14.9	12.8	7.8	4.3				
Annual mean temperature	26.2	26.2	24.1	20.1				
Hottest month mean maximum temperature	35.2	36.6	38.7	36.0				
Annual mean precipitation	1,190	906	311.3	241.0				
Driest quarter mean precipitation	0.1	0.0	2.5	27.5				
Wettest quarter mean precipitation	802.6	637.3	181.3	96.4				
Annual mean radiation	21.3	21.7	22.4	21.1				
Growth Index (microtherm plants)	0.00	0.00	0.01	0.04				
Growth Index (mesotherm plants)	0.19	0.14	0.05	0.08				
Growth Index (megatherm plants)	0.44	0.38	0.10	0.07				
Rainfall erosivity	10,200	7,620	1,531	595				
Annual mean “flow”	2.3	1.7			1.4	2.1		
C _v annual “flow”	0.7	1.2			1.9	0.8		
C _v monthly “flow”	2.7	3.8			5.8	2.7		
Maximum monthly “flow”	2.9	2.7			2.7	2.6		
Summer mean “flow”	1.9	1.5			1.2	1.3		
Autumn mean “flow”	1.5	1.2			1.1	1.3		
Winter mean “flow”	1.0	1.0			1.0	1.3		
Spring mean “flow”	1.0	1.0			1.0	1.1		
Snow contribution	0	0			0	0.7		
Catchment storage	1.7	1.9					1.4	0.2
Stream segment slope	1.8	1.9					1.0	3.6
Hillslope mean slope	2.6	2.8					1.5	5.7
Valley confinement	97.3	96.9					98	100
Relief Ratio	0.0	0.0					0.02	0.05
Elongation ratio	0.6	0.6					0.56	0.57
Catchment area	4.8	5.2					7	6
Surficial geology group 0	96.4	96.0						
Surficial geology group 1	1.6	1.6						
Surficial geology group 2	1.1	0.8						
Surficial geology group 3	0.0	0.0						
Surficial geology group 4	1.0	1.3						
Surficial geology group 5	0.0	0.3						
Lithology group 1	1.1	0.5						
Lithology group 2	0.0	0.0						
Lithology group 3	96.0	96.2						
Lithology group 4	0.0	0.0						
Lithology group 5	0.3	0.2						
Lithology group 6	0.8	2.1						
Lithology group 7	0.0	0.0						
Lithology group 8	0.0	0.0						
Lithology group 9	0.1	0.0						
Lithology group 10	0.0	0.0						

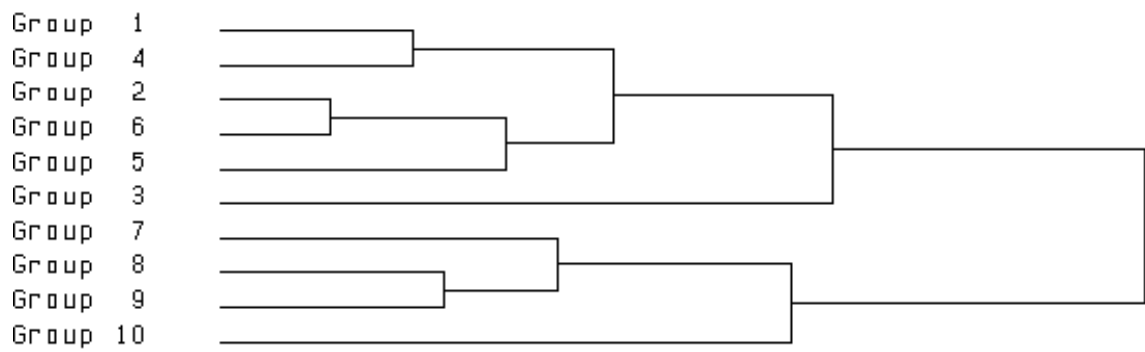
Broader groupings are apparent from the dendrograms that portray the hierarchical clustering of groups based on the inter-group dissimilarity matrix. At the highest level in the hierarchy, climatic attributes split streams into a warmer northern (groups 1 to 6) and colder southern divisions (groups 7 to 10) (Figure 7.15a). Group 10 (cold, alpine, sub-alpine) only joins other groups late in the fusion process. Similarly, group 3 is sufficiently different from other groups to remain on its own until the three groups level. The ordination plot (Figure 7.15b) confirms that the groups are quite distinct and occupy different sectors of the environmental space. All climatic attributes were strongly correlated with the ordination scores (r-squared values of 0.98 or better). The low estimate of stress (0.02) suggests the relationships between the groups are well defined and can be adequately summarised in three dimensions. Stress, a measure of the disparity between the input dissimilarities between objects and their distance in the ordination space, should generally be below 0.15 (Belbin and Collins, 2004).

Flow volume drives the primary division of streams in the flow classification (Figure 7.16). The larger rivers (groups 2, 7 and 10) form a distinct group. Flow variability produces a major split among the smaller to medium sized streams: high variability in groups 4, 5 and 6 and low to medium in the remaining groups.

At its highest level, the hierarchical clustering of the terrain groups (Figure 7.17a) partitions streams principally according to their degree of valley confinement. The ordination (Figure 7.17b) also shows groups aligned along an axis of increasing confinement in the ordination space. The three groups of streams (groups 3, 8 and 10) in unconfined valley settings are well separated from the remaining groups, themselves differentiated by their catchment size and degree of catchment storage. Local and catchment scale measures of energy availability (stream and hillslope gradients, relief ratio) produce the major division among the more confined streams.

For brevity, the dendrogram displaying the hierarchical clustering of the ALL groups is presented only from the 25 group level (Figure 7.18). Appendix 1 lists the ALL group members of these 25 meta groups. Climate and stream size are the major factors differentiating streams at the broadest scale. Streams are divided into 4 major groups: 1) hot and wet with a pronounced dry season; 2) hot and dry with high inter-annual flow variability; 3) cool to cold, moderately wet but without a marked dry period and 4) larger streams, warm to hot. Detection of patterns in the ordination is difficult because of the large number of groups and is not shown, although the ordination scores were used to derive the colours used to map the ALL groups in Figure 7.10.

a)



b)

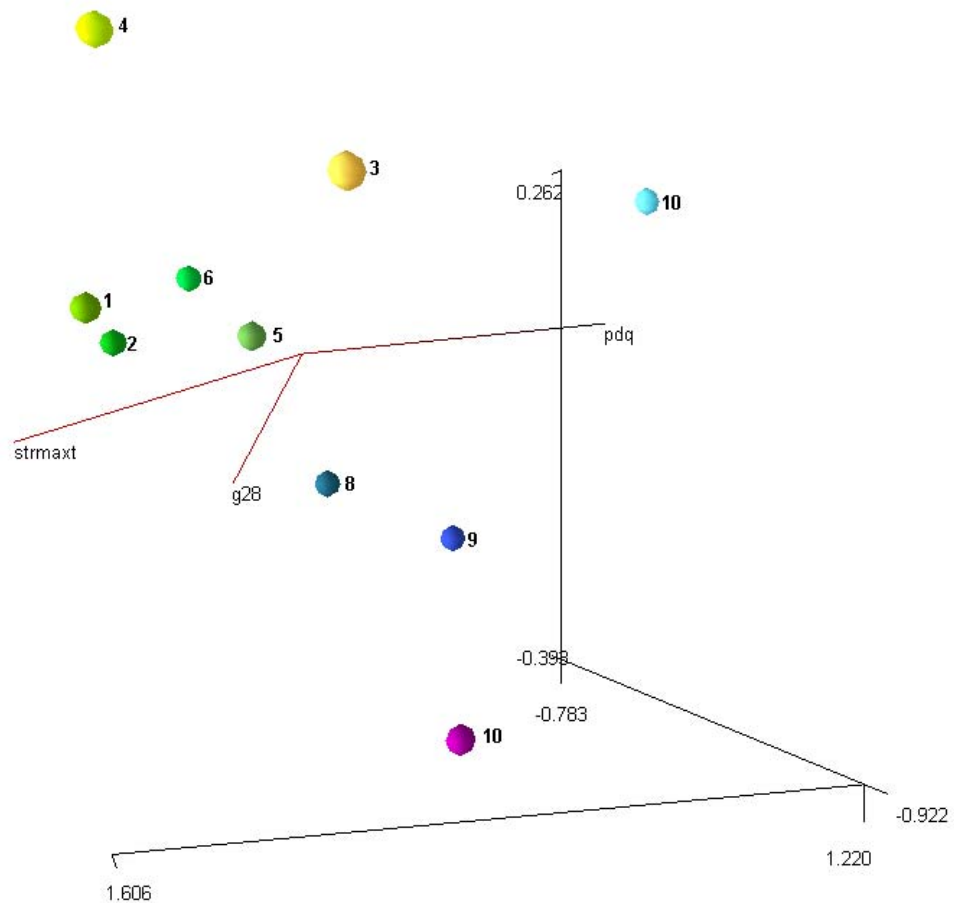
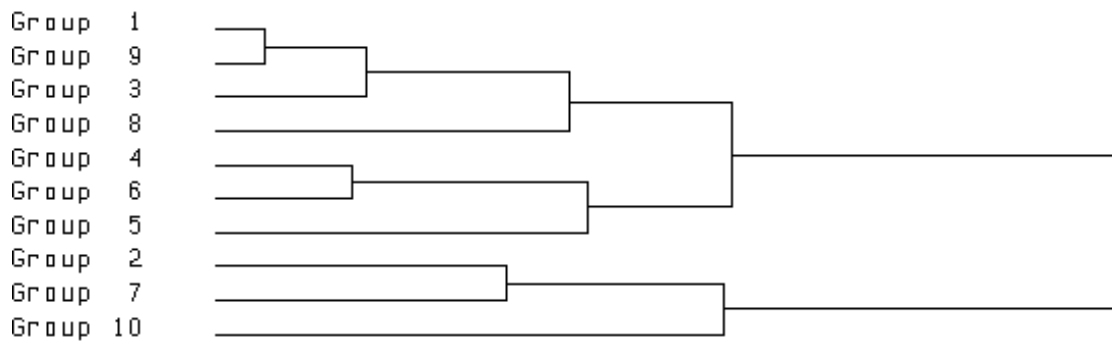


Figure 7.14 Relationship among climate groups: a) dendrogram showing hierarchical agglomerative clustering and b) 3-dimensional ordination. The ordination plot shows the vector of best fit in the ordination space, directed towards larger values of the climate attributes. Attribute labels: strmaxt= hottest month mean maximum temperature, pdq=driest quarter mean rainfall, g28=annual mean Growth Index (megatherm plants).

a)



b)

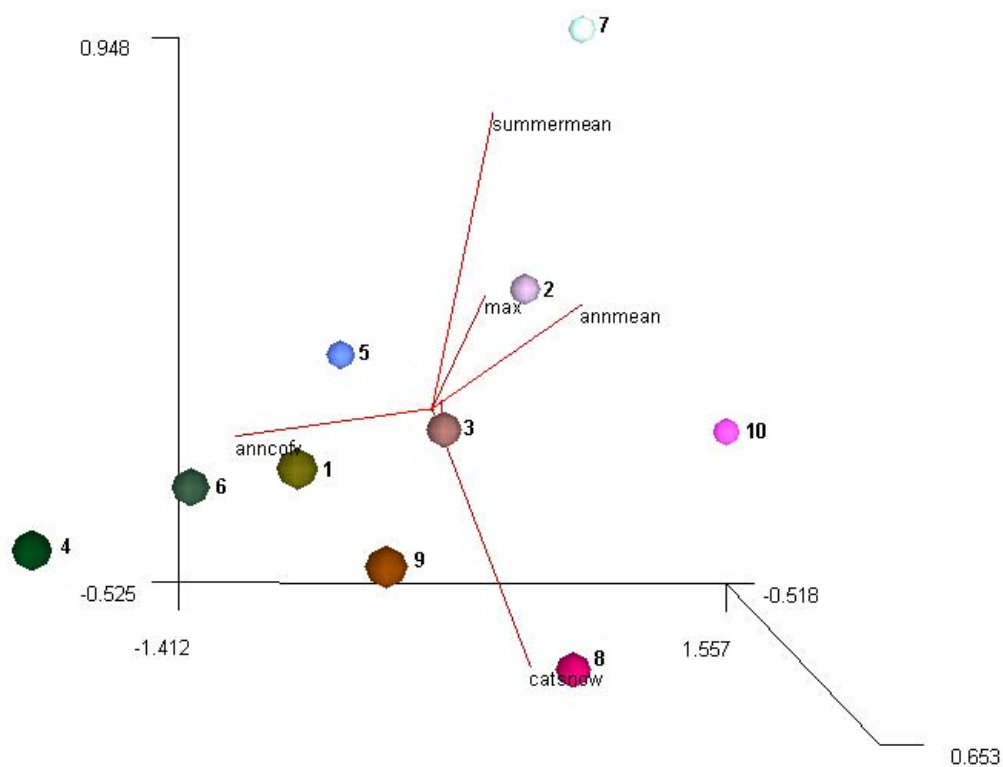
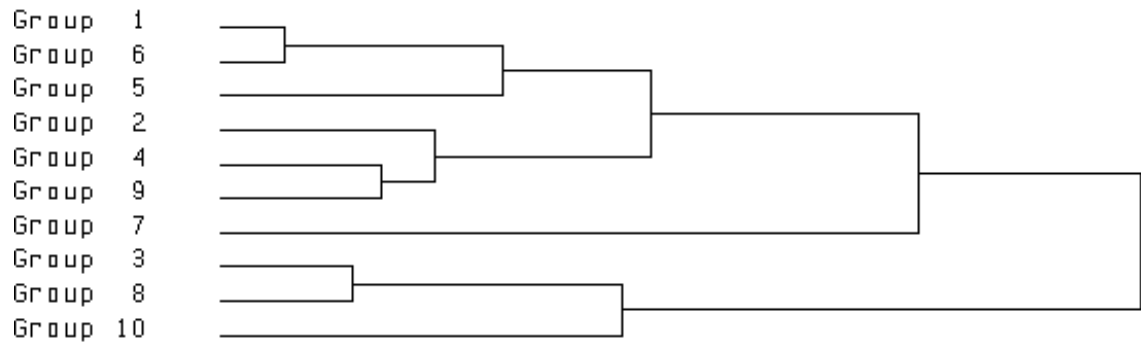


Figure 7.15 Relationship among flow groups: a) dendrogram showing hierarchical agglomerative clustering and b) 3-dimensional ordination. The ordination plot shows the vector of best fit in the ordination space, directed towards larger values of the flow attributes. Attribute labels: max=monthly maximum, annmean=annual mean, summermean=summer mean, anncofv=annual variability C_v , catsnow=snow contribution

a)



b)

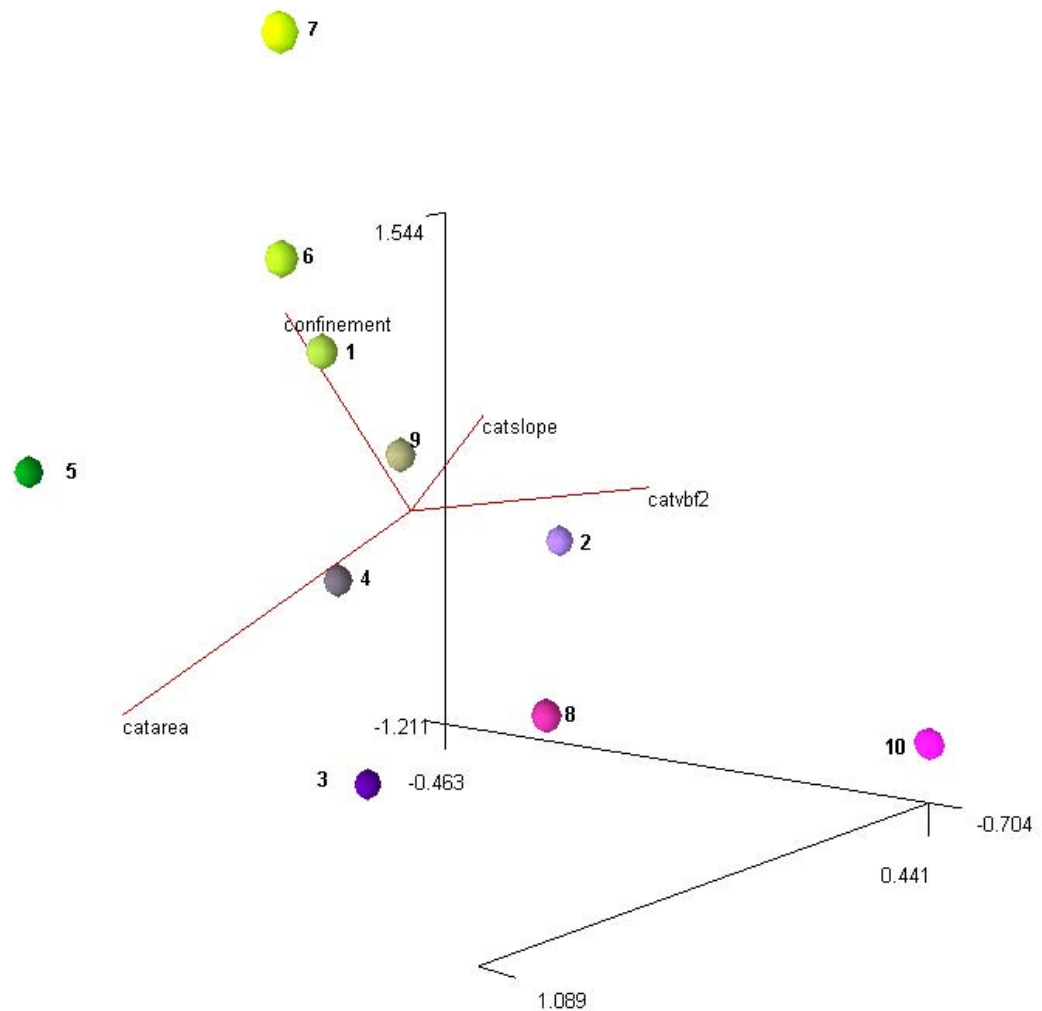


Figure 7.16 Relationship among terrain groups: a) dendrogram showing hierarchical agglomerative clustering and b) 3-dimensional ordination. The ordination plots show the vector of best fit in the ordination space, directed towards larger values of the terrain attributes Attribute labels: catarea= catchment area, catslope= hillslope, confinement=valley confinement, catvbf2=catchment storage

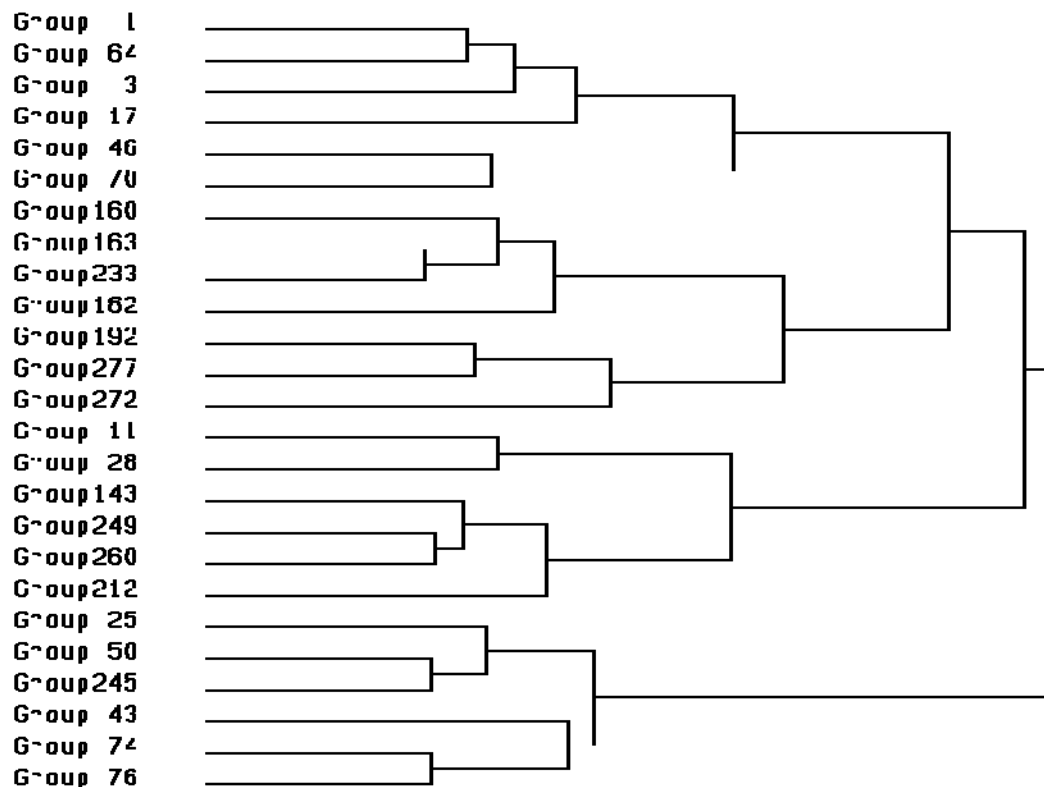


Figure 7.18 Relationship among the ALL groups. The dendrogram is truncated to the 25 group level. Group labels indicate the number of the first ALL group in the derived meta groups (Appendix 1)

7.3.2. Combining hierarchical levels

The hierarchical classification levels (“Levels 1 to 5”) were formed by combining the 10 group classifications as follows:

1. Level 1 – climate
2. Level 2 – climate + flow
3. Level 3 – climate + flow + geology
4. Level 4 – climate + flow + geology + terrain
5. Level 5 – climate + flow + geology + terrain + vegetation.

River Environment Types were thus defined by the combination of a stream segment’s group memberships. Groups were first labelled 0 to 9 by subtracting one from the group number so that the resulting River Environment Type codes may be treated as simple base-10 numbers. Not all of the possible River Environment Types exist among Australian streams (Table 7.15). Of the 100,000 Level 5 types potentially formed from the combination of the five classifications, just over 8,500 were found to occur. Moreover, many of these types are rare. More than 75% of the Level 5 or 57% of the Level 4 combinations have a total length of stream of less than 100km while many are found in just one segment. The size of the groups varies by

Table 7.15 Summary properties of the classifications of River Environment Types.

Property		Classification					
		ALL	Level 1	Level 2	Level 3	Level 4	Level 5
No. types		355	10	81	420	2,361	8,614
No. types with aggregate length < 100km		0	0	9	151	1,357	6,620
No. segments of each type	Min.	103	19,674	1	1	1	1
	Mean	2,692	95,563	11,798	2,275	405	111
	Max.	39,836	203,923	169,108	122,132	52,451	23,406
Aggregate length of segments of each type (km)	Min.	147	56,181	<1	<1	<1	<1
	Mean	7,904	280,586	34,640	6,681	1,188	326
	Max.	114,038	625,908	495,993	355,656	147,812	64,803
Aggregate area of segment sub-catchments of each type (km ²)	Min.	194	64,441	<1	<1	<1	<1
	Mean	16,570	588,251	72,624	14,006	2,492	683
	Max.	151,231	1,615,398	1,183,643	813,264	205,295	95,791

an order of magnitude, for example, more than 10 times the length of stream in the smallest Level 1 group (Climate group 10, Level 1 River Environment Type 9) falls within the largest group (Climate group 2, Level 1 River Environment Type 1) (Table 7.15). In contrast, none of the ALL groups has less than 100 members or total less than 100km in length (Table 7.15). Nevertheless, there is a large disparity in the aggregate area of the segment sub-catchments among the groups of the ALL classification. The largest ALL group (Group 158), encompassing streams across much of the southern section of the arid zone, occupies an area more than 750 times that of the smallest (Group 355), found in south-west Tasmania.

7.3.3. Numbers of groups

Alternative numbers of groups were assessed for the climate, flow and terrain classifications. Plots of the average dissimilarity of each segment from its group centroid (average deviation) against the numbers of groups indicate that the homogeneity of the groups would increase appreciably with a finer partitioning (Figure 7.19). In each case, there is a sharp decline in the average deviation as the number of groups increases, with an apparent change in the slope of the relationship between 20 and 30 groups. The modified measure of classification strength also

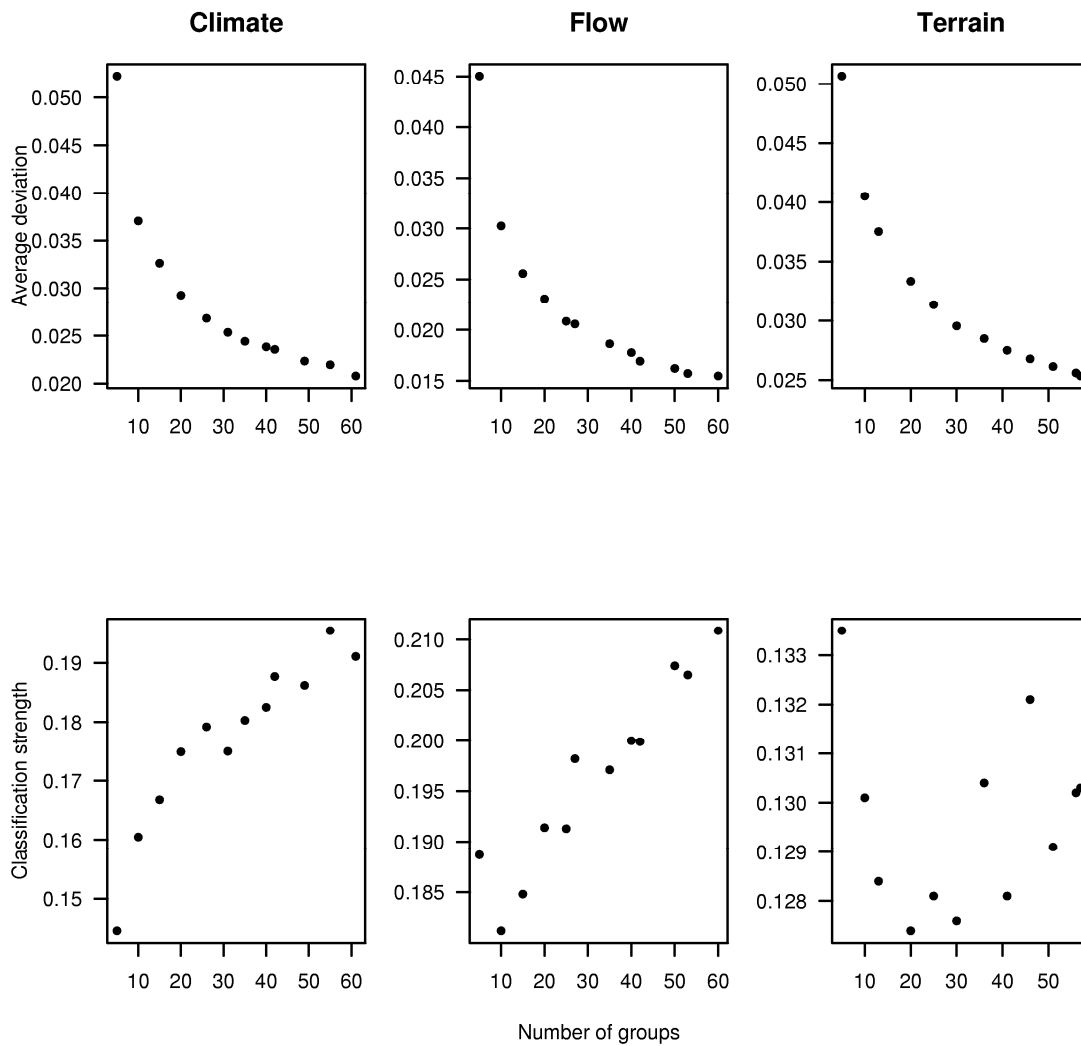


Figure 7.17 Comparison of the average deviation and modified classification strength for alternative numbers of groups for climate, flow and terrain classifications. Classification strength is expressed as the absolute difference $\bar{W} - \bar{B}$ computed from respectively, the within and between group distances of segments to the group centroids.

increases quite sharply with increasing numbers of groups for both the climate and flow classifications (Figure 7.19), although the absolute value of the improvement in CS is not great. There is some evidence of a break in the slope of this relationship, again between about 20 and 30 groups though the change in slope is less marked than for average deviation. In contrast, there is no clear trend in classification strength with increasing numbers of terrain groups, with several local maximum and a global maximum around 45 groups (Figure 7.19).

7.4. Discussion

Stream segments were individually characterised by landscape attributes that play a critical role in controlling riverine ecosystem processes. Classifications summarise and integrate the spatial patterns in the distribution of these attribute values. Broad spatial patterns are most evident in the climatic attributes. The climate classification is strongly influenced by the north-south

temperature gradient with secondary gradients due to elevation and distance from the coast. These patterns also come through strongly in the combined ALL classification, superimposed by secondary divisions primarily reflecting the variation in flow volume. Thus, the primary divisions amongst the ALL groups parallel the hierarchical combination of the 10 group classifications. Patterns in the spatial distribution of flow, geology, terrain and vegetation groups are more complex and not easily ascribed to broad regions although a single bedrock geology group (sedimentary rocks) dominates much of the continent.

While the classifications are the primary objective, the continent wide attribute maps themselves generate interesting insights. For instance, the extreme variability of runoff of Australia's rivers and streams and its role in shaping the adaptations of the aquatic biota is often highlighted (Boulton and Brock, 1999) but the modelled catchment water balance suggests there are also many rivers with quite predictable flows.

There are no other continental classifications of Australian streams to which the landscape

Table 7.16 Correspondence between Level 1 (Climate) groups and the agroclimatic classification of Hutchinson and colleagues (not incorporating bioregional boundaries) (Hutchinson *et al.*, 2005).

Agroclimatic class	Proportion of grid cells in Level 1 group in agroclimatic class (%)									
	1	2	3	4	5	6	7	8	9	10
B1									1	18
B2										35
D5							<1	4	72	47
E1								9	16	
E2					1	5	<1	32	1	
E3							6	22	6	
E4					52	4	<1	17		
E6						34		15		
E7			16		6		17	1		
F3							21	<1	4	<1
F4			5				55		<1	<1
G	1	83				57				
H	32	17			4	<1				
I1	3		7	84	1					
I2	63	<1		11						
I3	1		49	1	36		<1			
J1			20	4						
J2			3							

classifications can be directly compared. The only other classification with continental coverage is the fish biogeographic provinces of Unmack (2001) but these drainage basin groupings do not distinguish sub-basin scale variability among streams. There are clear similarities between the 10 group climate classification that forms Level 1 and the 18 group agroclimatic classification of Hutchinson and colleagues (2005). Most of the Level 1 groups correspond with just one or two of the agroclimatic classes (Table 7.16).

7.4.1. Classification approach

The environmental domain approach of grouping streams by their shared similarity, computed across all attributes collectively, created a single classification (ALL). However, conflicting spatial patterns in the distribution of attribute values (e.g. sub-catchment versus catchment scale, categorical substrate versus continuous climate and terrain) produced relatively heterogeneous groupings. In particular, stream segments with a wide range of substrate characteristics are grouped together. The environmental characteristics of the River Environment Types formed by combination of separate landscape classifications were more readily ascribed. The hierarchical classification is also more easily tailored to suit the applicable scale of the conservation planning task. However, to incorporate the influence of the same landscape controls, it defines a much larger number of River Environment Types than ALL.

To understand why this occurs consider the following example. Assume two stream segments share virtually identical values on all attributes except two of the flow attributes. The differences in the values of these two attributes are sufficient to classify the segments into disparate groups in the ten group flow classification. Because the flow attributes are just one of four attribute groups in the ALL classification their contribution to the overall measure of association between the segments is just 25%. The similarity of the values across the other attribute groups overwhelms the differences in flow values and ALOB groups the segments together. Hence, the ALL classification produces just one River Environment Type where the hierarchical produces two.

7.4.2. Systematic conservation planning requirements

The river environment classifications address the requirements for systematic conservation planning (Section 7.1). They embrace the whole channel network including both large and small streams. The objects of the classification are spatially referenced so the classifications are readily mapped. The use of repeatable clustering methods addresses the need for consistently

and transparently derived groups. The classificatory strategy is explicitly defined so that choices can be critically reviewed.

The classifications are based on temporally stable attributes that describe long-term landscape controls on ecosystem processes. They produce groups indicative of potential states (Frissell *et al.*, 1986) independently of any human activities that may influence current conditions. In contrast to classifications based on highly dynamic properties of river systems (e.g. channel geomorphic characteristics or many biotic communities), it will not be necessary to frequently revise the River Environment Types.

The river environment classifications enable application at multiple scales. The hierarchical combination of classifications derives from commonly accepted understanding of the relative spatio-temporal scale of influence of the principal controlling factors (Naiman *et al.*, 1992; Maxwell *et al.*, 1995; Cleland *et al.*, 1997; Montgomery, 1999; Snelder and Biggs, 2002). Climate, widely acknowledged as the primary driving variable of ecosystem patterns and processes, operates at broad spatial scales (Bailey, 1996; Cleland *et al.*, 1997; Calvert *et al.*, 2001; Snelder and Biggs, 2002) and is thus the basis of the highest level sub-division of stream segments. The basic elements of the flow regime are incorporated via the catchment water balance, the “flow” level in the hierarchical classification. Although a function of climatic parameters (rainfall and evapotranspiration) the catchment water balance is included separately because it specifically integrates the influence of climate across the catchment. Flow regime plays a fundamental role in shaping the evolution of aquatic communities and ecological processes (Naiman *et al.*, 2002). Geology is the other independent control on riverine ecosystem character (Knighton, 1998) and so forms the next level in the hierarchical classification. Topographic and vegetation controls operate over shorter time scales and smaller spatial scales within the constraints imposed by historical climate and geology (Frissell *et al.*, 1986 ; Naiman *et al.*, 1992) and accordingly these factors are the basis of the lowest levels in the hierarchy. Although not explicitly incorporating the hierarchical relationships between controlling factors in the same way, the ALL classification may be applied at a range of scales by extracting new groupings from the higher order relationships among groups that are displayed in the dendrogram.

7.4.3. Opportunities for improvement

The river environment classifications address many of the requirements for conservation planning. Chapter 8 will provide evidence that they are also ecologically relevant. Nevertheless, there are opportunities for improvement. Future revisions should focus on the hierarchical organization and degree of partitioning and reducing the uncertainty associated with the attribution of critical landscape factors.

7.4.3.1. Hierarchical organization

The landscape classifications reflect the relative scale of influence of river ecosystem controls, but do not conform strictly to the nested hierarchical structure of the stream system (Section 2.3). This is because the classification employs a “bottom up” approach, grouping stream segments according to their shared similarity rather than a “top down” one of successively subdividing drainage basins. Headwater segments, for example, are usually grouped with other headwater segments in adjoining drainage basins rather than downstream segments in the same basin. From an ecological and functional perspective this makes sense.

Similarity is measured across attributes that describe landscape characteristics at multiple spatial scales. In the case of climate, geology and vegetation this is the stream segment and its immediate sub-catchment. The flow classification is based on attributes that integrate runoff processes across the catchment while the terrain and ALL classifications combine attributes at both spatial scales. By integrating attribute values in the upstream contributing area, the catchment descriptors provide a mechanism for explicit recognition of linkages between elements of the drainage network. Still, the mixing of scales may explain the heterogeneity within groups of the terrain and ALL classifications. Segments that share similar values of attributes at one scale may have little in common across attributes measured at different scales.

Classification of hydrologic units at a range of spatial scales, not just the segment scale, would better represent the hierarchical structure of the drainage system, but not retain the relationships between the nested units. Segments, sub-catchments and drainage basins would be independently classified with attributes applicable to each scale, but units grouped at one scale may end up in different groups when the larger spatial units to which they belong were classified. Such conflicts could create confusion for users of the classification.

Classifying spatial units according to the group membership of their component lower level units offers an alternative approach to multi-scale classification. This is premised on a view that characteristic associations of river types form recognisable river landscapes at a range of spatial scales. With this approach, the similarity of the landscape groupings of their segments would be the basis for the clustering of sub-catchments. In turn, patterns in the groupings of these sub-catchments would underpin the classification of larger sub-catchments and so on up to drainage basins. This approach is appealing because it maintains the nested hierarchy of spatial units while explicitly recognising linkages among elements of the drainage system.

While a suitable spatial framework is available (the nested catchments, see Chapter 5), appropriate means of attributing catchment units may be harder to identify. Proportional areas might be used to code the group membership of lower level units in the same way that geology classes were attributed to stream segments for the ALL classification. However, such coding treats each group independently for the purposes of calculating the similarity between catchment

units. It does not take advantage of the inter-group dissimilarity matrix so conveys no information about the relationships between groups. Moreover, it potentially produces a large and sparse data matrix (dependent upon the number of groups). Because the similarity between objects is less accurately estimated when objects share little in common (Belbin and Collins, 2004), measures of association between catchment units may be unreliable. Data reduction methods such as Principle Components Analysis (PCA) (Digby and Kempton, 1987) or more recent variants that cater for categorical attributes (Vaughan and Ormerod, 2005) might help in this regard. Even so, the performance of completely automated methods of pattern analysis may not be able to match that of manual interpretation at the broadest levels of generalization (MacMillan *et al.*, 2004).

7.4.3.2. Attribute uncertainty

Characterising landscape processes

The effectiveness of the river environment classifications depends on the ability of the classifying attributes to capture the critical landscape controls on stream ecosystems. Weak differentiation of landscape controls by river environment classifications may be the consequence of omission or poor characterization of important physical processes.

Present day climate and geology mapping does not necessarily reflect the role played by historical climatic and geological processes in shaping modern rivers and their biota (Collier and Allibone, 2001; Ollier, 2001; Jerie *et al.*, 2003). Unfortunately, relevant mapping, for example, of areas affected by the most recent and maximum glaciation (Jerie *et al.*, 2003), with consistent continent-wide coverage at appropriate spatial scales is lacking. The soil properties that influence runoff and erosion processes and the chemistry of runoff waters might also have been included, but limitations in the available spatial data (e.g. National Land and Water Resources Audit, 1999) would have introduced unacceptably high levels of uncertainty at the scale of individual stream segments. The ongoing development of the Australian Soil Resource Information System (ASRIS) (Johnston *et al.*, 2003) should assist the inclusion of soil information into future river environment classifications.

The simple catchment water balance was computed from monthly rainfall totals rather than actual rainfall events and does not account for transmission losses in streamflow routing. Contributions to streamflow from surface (overland and interflow) and groundwater are not partitioned yet in many rivers, the exchanges between surface and sub-surface waters have a crucial function in regulating water temperature and suspended sediment loads and in the exchange of nutrients between streams and riparian or floodplain systems (Jensen, Goodman, Frissell *et al.*, 2001; Amoros and Bornette, 2002; Malard *et al.*, 2002). Not surprisingly, the

monthly water balance poorly simulates low flows, normally derived from groundwater discharge (Smakhtin, 2001).

Still, the river environment classifications identify groupings of streams that are relatively homogeneous in other landscape factors (climate, geology, topography, and vegetation) that broadly indicate the spatial extent and influence of groundwater flows (McMahon, 1977; Salama *et al.*, 1997; Lacey and Grayson, 1998; Jensen, Goodman, Frissell *et al.*, 2001; Marani *et al.*, 2001). Variation in other ecologically important facets of flow not reflected by the monthly catchment water balance (e.g. the rate and magnitude of change of flow Puckridge *et al.*, 1998; Grouns and Marsh, 2000; Olden and Poff, 2003) are also largely controlled by these landscape factors. There are exceptions. Complex surface and groundwater interactions may be independent of topographic controls (Devito *et al.*, 2005). Lateral flows in rock joints and fissures, for example, may cross topographically defined drainage divides (Halwas and Church, 2002; McIntosh *et al.*, 2005). Nonetheless, the river environment groups are expected to exhibit generally similar hydrological responses at a range of temporal scales (notwithstanding concerns about data quality). This hypothesis will be tested for an extensive set of gauging stations in collaboration with Griffith University's Riverine Landscape Unit in a Land and Water funded project beginning in 2006.

Categorical variables

Categorical map data such as lithology or vegetation are difficult to attribute in ways that suitably reflect their wide ranging influence on stream ecosystem characteristics. Even where quantitative ratings can be assigned to individual categories (e.g. Nix, Stein, Stein *et al.*, 1992) there remains the problem of integrating these values across the catchment. Summary statistics (e.g. means or ranges) are uninformative and may be misleading. Modelling the physical processes that shape the character and behaviour of stream ecosystems offers a theoretical solution to this dilemma. For instance, the breakdown and transport of materials as they influence the nature of the stream substratum might be modelled as a function of catchment lithology. Unfortunately, the coarse scale of national geological mapping currently precludes such an approach. Illustrating the degree of uncertainty that can arise, the NLWRA estimates that true pre-European hillslope erosion values might be as much as 50 times more or less than those predicted using the available continental scale soil and geology mapping (National Land and Water Resources Audit, 1999).

Two alternative solutions to the problem of attributing categorical variables were employed. The first, attributing stream segments with the areal proportion of each of the substrate categories within the catchment, was tested for the combined (ALL) classification. This approach enabled segments to be grouped according to their similarity across a broader range of substrate attributes (bedrock lithology and surficial cover) but provides no indication of the

relative position or influence of rock types within the catchment. Moreover, it generates a large number of attributes often with many zero values (i.e. for each substrate category not occurring within the catchment). Commonly used measures such as that of Bray and Curtis exclude zero-zero matches from the calculation of dissimilarity (Belbin, 1993b), but as noted above, association measures are less reliable when objects share little in common. Alternative weightings of attributes might have produced groups with more homogenous substrate conditions but experimenting with the relative weighting was precluded by the long computing time needed to generate each classification. It would be feasible for a smaller subset of river segments.

For the hierarchical classification, segments were simply attributed with the dominant lithological category in the segment sub-catchment. This is equivalent to the commonly adopted method of defining relatively homogenous regions within which to apply further classification (Kondolf *et al.*, 2003). However, sub-dominant rock types may also be important (Snelder and Biggs, 2002), so attributing a single category is rarely reasonable in large, heterogenous catchments. Accordingly, the dominant category was applied only for the segment sub-catchment. The serious disadvantage of this method is the failure to take account of the influence of the broader catchment.

Spatial data resolution

The need for consistency dictated the use of spatial data layers with continental coverage. Unfortunately, the only data available for some attributes was sourced from small (coarse) scale mapping that may not support the 1:250,000 scale of the underlying spatial framework. This is particularly so for the geology mapping used. Produced at a scale of 1:2,500,000 the boundaries of mapping units on the Geology of Australia map are highly generalised. Map units represent the dominant lithological type yet minor geological formations may play an important role in shaping stream properties. The weathering products of sedimentary rocks, for instance, have a disproportionate influence on stream chemistry (Meybeck, 1987 cited in Knighton, 1998).

While geological mapping in electronic form is available for much of the country at 1:250,000 scale or finer it would be a major undertaking to rectify the many inconsistencies that exist in the coding and lithological description of map units across state or map sheet tile boundaries. The NLWRA recently compiled the best available geological mapping into a consistently classified coverage, assigning map units to one of 23 categories of significance for soil formation (National Land and Water Resources Audit, 2001e), but this task extended only to the Intensive Landuse Zone. Even so, there remain obvious coding discrepancies at State or map tile borders. A new national geological map is currently under development by Geoscience Australia at 1:1 million scale. As of September 2005, coverage of outcrop geology for the

eastern states of Australia was available

(<http://www.ga.gov.au/ausgeonews/ausgeonews200509/productnews.jsp>).

Uncertainty also results from the overlay of data of different spatial scales (Bailey, 1988; Gaeuman *et al.*, 2005). Combining the coarse scale geological mapping (1:2.5 million scale) with much finer scale (1:250,000) climatic and topographic data will have produced spurious combinations of attributes and artificial groupings that have no ecological significance. To detect such artefacts, independent higher resolution data sources or field surveys would be required. Once available, the new seamless geological coverage at 1:1 million scale would undoubtedly improve future revisions of the river environment classifications, but scale mismatch, though less extreme, will continue to confound classifications. Whether the information gains that arise from the inclusion of lithological attributes outweigh the increased uncertainty they produce is an important research question to follow up.

7.4.3.3. Number of groups

The number of groups in the classification, and hence the number of stream types recognised, will influence conservation planning outcomes including evaluation of the representativeness of existing protected areas (Pressey and Logan, 1994). More homogenous groups derive from a finer partitioning and are likely to provide better targets for representation of biodiversity in a protected area system (Pressey and Logan, 1994), a more accurate description of reference conditions for assessment of ecological condition and a reliable basis for extrapolation of survey results. Yet, too many groups will complicate management prescriptions and be more difficult to interpret and communicate.

Uncertainty in the attribute data limits much finer partitioning of stream segments in the ALL classification. At 355 groups, the differences in attribute values between the closest groups are approaching their estimated error (Table 7.14). This is clearly not the case, however, for the 10 group climate, flow and terrain classifications. Both the plots of average deviation and the modified classification strength indicate a finer partitioning would substantially reduce the heterogeneity of the resulting groups though neither readily identifies the best classification. A good classification reflects natural structure in the data (Bao, 2000) but natural partitions are not always readily apparent. While breaks coinciding with sudden changes in river character and behaviour at confluences (Benda *et al.*, 2004) or abrupt changes in valley confinement (Brierley and Fryirs, 2002) may be easily recognised, they are less perceptible when variation occurs continuously, reflecting change that is more gradual (Brierley and Fryirs, 2002).

There are no widely accepted criteria to objectively determine an optimal number of clusters (Belbin, 1993a). Furthermore, different indicators may not agree on this number (Mufti *et al.*, 2005) so other evidence, including expert knowledge, is important to assist evaluation. From the

analysis of alternative partitions of the climate, flow and terrain classifications (Section 7.3.3), it appears that a number of clusters between 20 and 30 might better balance the desire for within group homogeneity with the need for parsimony. Still, the lack of independence among these factors (each being a derivative of the DEM in some form) complicates any decision. Though more complex, analysing the adequacy of the level of partitioning of the classifications in combination would enable more robust conclusions.

7.5. Conclusions

For the first time, the rivers and streams of Australia have been consistently characterised continent-wide, producing a comprehensive picture of variation in landscape controls on aquatic ecosystem patterns and processes, both within and between drainage basins. The classifications of stream segments summarise these patterns at multiple spatial scales, providing a potentially useful landscape framework for many conservation planning applications. First however, their ecological relevance must be validated (Chapter 8).

The classifications group hydrologic units according to their similarity in environmental data space. Consequently, group members may be widely dispersed geographically. The attributes selected to characterise landscape controls on stream systems reflect the important linkages between the stream and its catchment. Classes derive from the structure within the attribute data, rather than being imposed *a priori*.

Outcomes depend on the classification approach adopted. The environmental domain approach classified stream segments more parsimoniously (i.e. fewer groups) but groups tended to be less homogenous and therefore more difficult to interpret than the hierarchical classification. The number of types is an important property of a classification with implications for conservation planning applications. For a larger number of River Environment Types more rivers may be required to establish a representative system of protected rivers (Chapter 9). Acquiring sufficiently representative survey data to evaluate the ecological significance of the landscape classifications also becomes more difficult with increasing numbers of river types (Chapter 8).

Chapter 8. Are the landscape classifications ecologically relevant?

The development of two landscape classifications for Australian rivers and streams was described in Chapter 7. The classifications were shown to address many of the requirements of systematic conservation planning. Here, the ecological relevance of the classifications are considered, judged by their ability to represent variation in stream biota (fish and macroinvertebrates) and habitat characteristics at sites from two river surveys: the NSW Rivers Survey (Harris and Gehrke, 1997b) and the AusRivAS National River Health Survey (Smith *et al.*, 1999; Turak *et al.*, 1999). Understanding of the variability among river environment groups is also assisted by exploratory analysis of the habitat characteristics recorded at 22 sites on rivers across the Northern Territory.

The methods section of this chapter (Section 8.1) begins with a description of the survey data and its preparation. It follows with an explanation of the test statistics used to gauge classification performance including a measure of classification strength based on biotic similarity and explanatory power in mixed effect models fitted to site physical and chemical variables. For comparison, the IBRA regionalisation (Environment Australia, 2000) that underpins the development of the National Reserve System is also assessed against the same test data. The test results are reported next in Section 8.2. The discussion (Section 8.3) compares the performance of the river environment classifications and the IBRA regionalisation. The classification strengths of the river environment classification are also compared to that reported for landscape classifications in international studies. While the global tests validate the ecological relevance of the river environment classifications, they mask considerable variability in performance among the groups. Possible explanations for this variability conclude the discussion.

8.1. Methods

8.1.1. Test data

8.1.1.1. The NSW Rivers Survey

NSW Fisheries provided data from the NSW Rivers Survey, a systematic survey of the distribution and abundance of fish in NSW rivers (Harris and Gehrke, 1997b). The data include records from the original survey conducted over three years from 1994 to 1996 (Harris and

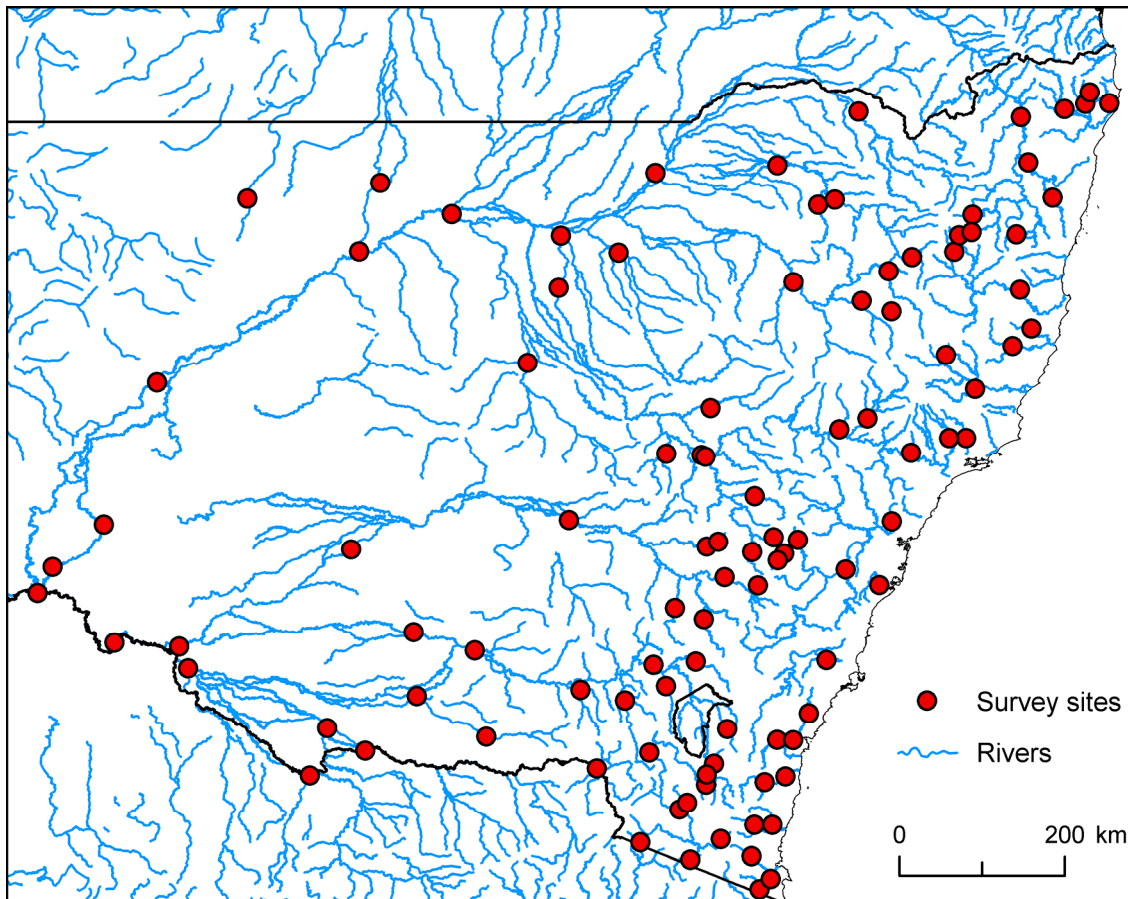


Figure 8.1 Distribution of sites in the NSW Rivers Survey. Five sites were omitted from the analysis of biotic similarity because they were highly disturbed or no native fish species were observed at the site.

Gehrke, 1997a) and additional surveys between 1997 and 1999. The data record the abundance of 58 fish species observed at 102 sites across NSW, sampled on between one and six occasions (Figure 8.1). Sites were selected using a stratified random sampling scheme to represent broad freshwater ecological regions (Murray, Darling, North Coast and South Coast) and three topographic classes (montane, slopes, lowland) (Harris and Gehrke, 1997b).

A sites by species abundance matrix was assembled by averaging the total catch over all surveys. Alien and estuarine species were omitted. The three sites on the Cockburn, Goulburn and Castlereagh Rivers that had been dropped from the survey because they were highly disturbed (Harris and Gehrke, 1997b) were removed from the database. Following Gehrke (1997), abundance data was fourth root transformed for analysis. It was also converted to a presence/ absence matrix by altering all non-zero abundance values to ones.

8.1.1.2. AusRivAS River Health Survey

The Australian River Assessment Scheme (AusRivAS) is a national system to assess river health using rapid bioassessment procedures (Davies, 1994; Barmuta *et al.*, 1998; Smith *et al.*, 1999; Turak *et al.*, 1999). River health at a test site is inferred from the differences between observed populations of aquatic macroinvertebrates with that predicted for the site under natural

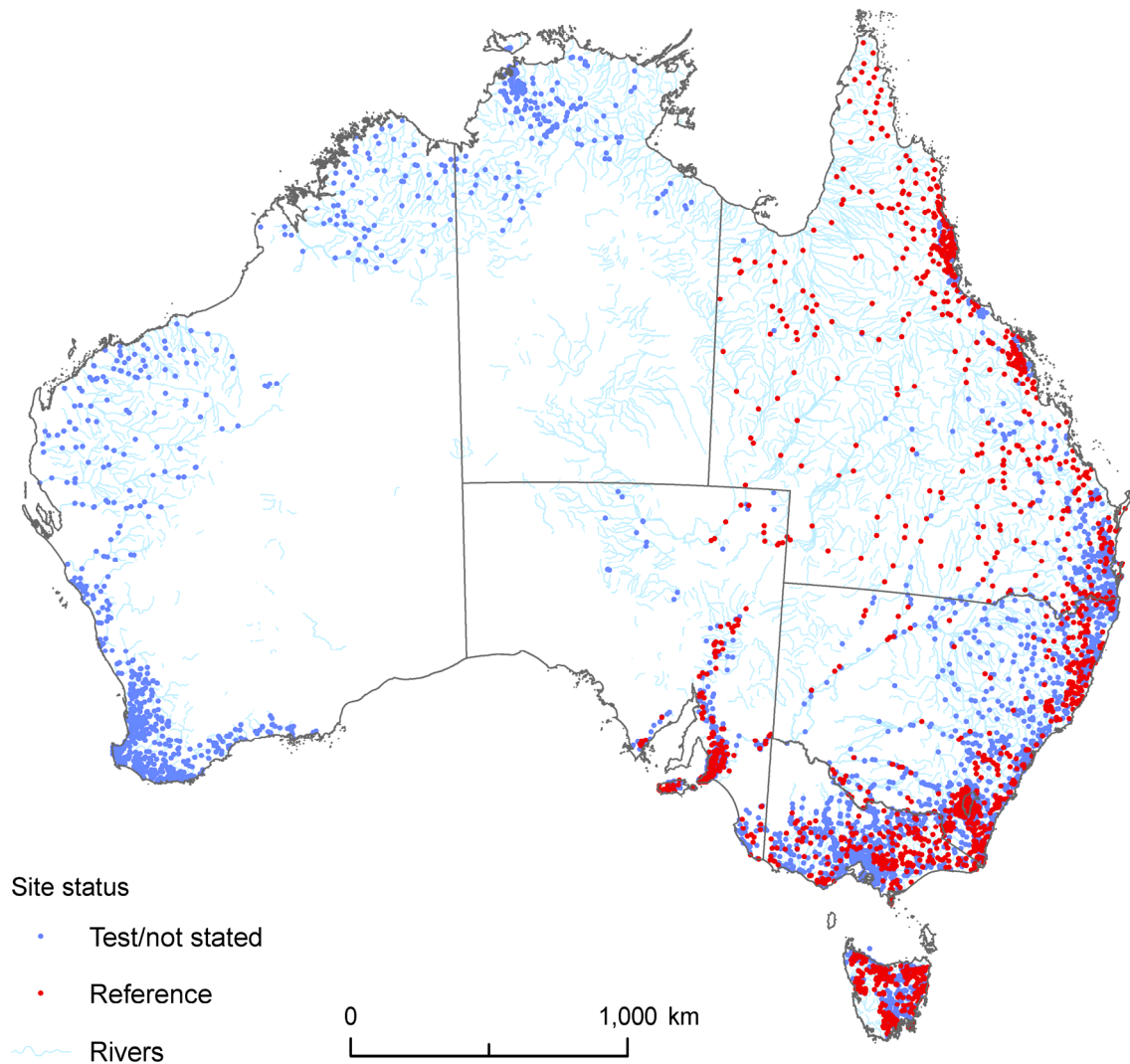


Figure 8.2 Distribution of AusRivAS sites.

conditions. Predictive models, based on the physical, chemical and riparian vegetation characteristics of a site, are built from data collected at a large number of reference sites using standard sampling protocols (for details see <http://ausrivas.canberra.edu.au/Bioassessment/Macroinvertebrates>). Reference sites are located on rivers minimally impacted by human activities wherever possible or elsewhere “best available” (Davies, 1994).

The AusRivAS National River Health database is the most comprehensive and consistent continent-wide survey of the ecological characteristics of Australian rivers and thus forms a valuable resource for evaluating the river environment classifications. A copy of the database, current to October 2004, was supplied by the Commonwealth Department of Environment and Heritage. It contains counts of macroinvertebrates, identified to family level, and matching site variables recorded at sites nationwide (Figure 8.2). Most sites had been visited on more than one occasion between 1994 and 2003 for a total of 14,909 visits to 4944 sites. Macroinvertebrate

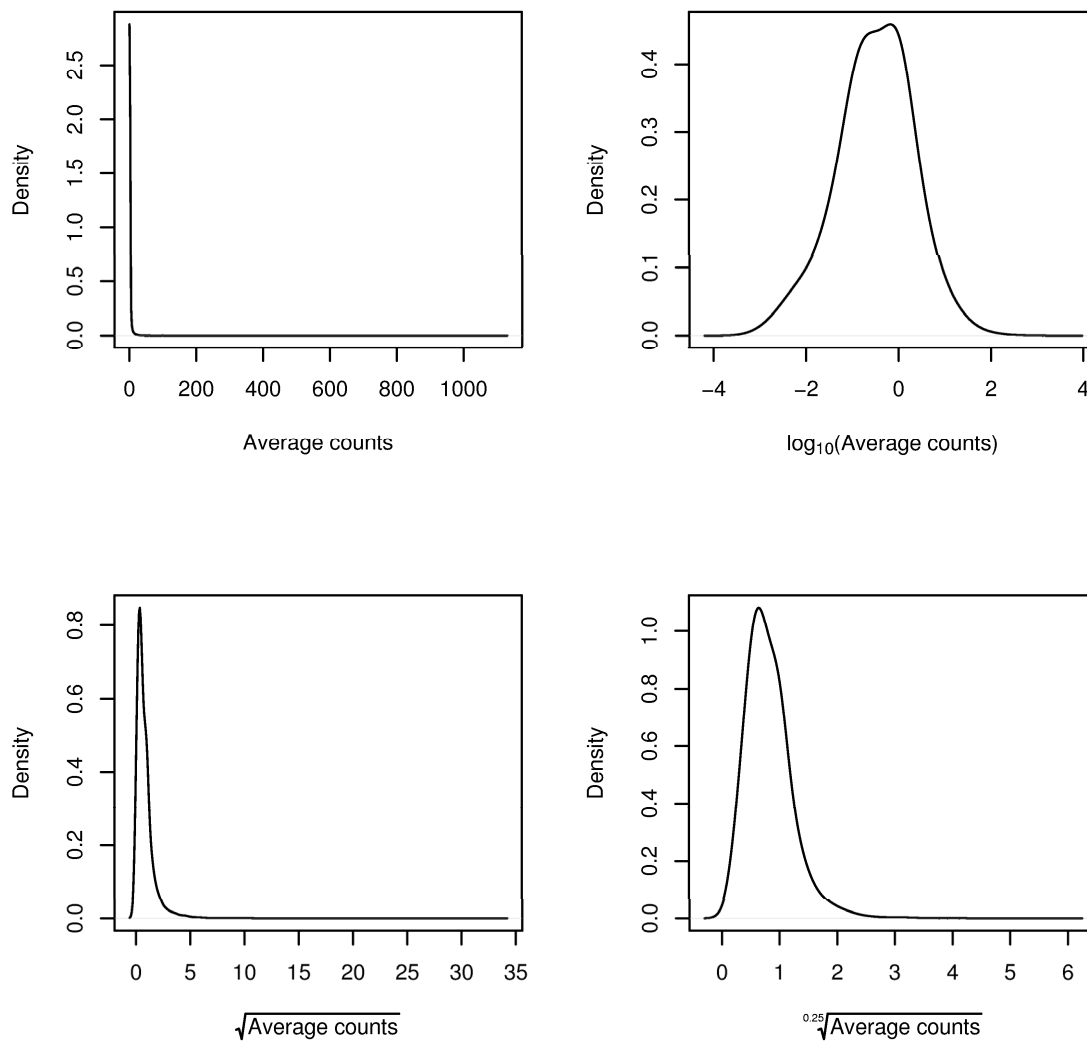


Figure 8.3 Distribution of macroinvertebrate count values showing effect of different data transformations. The $\log_{10}$ transformation was used as it most effectively reduced the data skew.

counts and associated site variables were recorded separately for specified instream habitats (e.g. edge or riffle).

Counts of macroinvertebrates were extracted from the Microsoft Access database, averaged across all visits and habitats and assembled into a matrix of sites by taxa. To reduce the chances of confounding inter-group differences with anthropogenic impacts, only sites labelled as reference were used. Counts were $\log_{10}$ transformed to reduce the substantial skew (Figure 8.3) and converted to presence / absence by changing all non-zero values to ones.

Physical and chemical variables were also extracted for sites attributed as “reference” (Table 8.1). Because the status of Western Australian sites (i.e. reference or test) was not identified in the database, data records with a river health rating of Band A (i.e. “similar to reference”) based on their habitat Taxa O/E scores (Barmuta *et al.*, 1998) were also included. These site variables describe characteristics of the stream habitat strongly associated with the distribution of macroinvertebrate taxa (Davies, 1994) but are also more generally descriptive of stream habitats (Parsons *et al.*, 2004b). Variables recorded at relatively few sites (< 150 sites), such as those

describing the cover by different macrophyte types, were excluded. Similarly excluded were categorical variables and variables entered with indicative values only (e.g. total phosphorous < x mg/l).

The AusRivAS sampling protocols aimed at consistency across the states and territories. The data were assembled into a single national database, but inconsistent coding or recording of variables was still noted. The distance from source field, for instance, appeared to have values entered in metres for sites in NSW and Tasmania and kilometres elsewhere. Different codes (e.g. blanks, “555”, “9999”, “-9999”) were used to indicate missing data. Similarly, habitats were entered with a variety of descriptors (e.g. “Edgewater”, “Edge”, “E”, “edge”, “EdgeWater” for edge habitats). To identify potentially incompatible data, summary statistical profiles of each attribute were produced. Inconsistent coding was rectified wherever possible with reference to the field survey manuals used in each jurisdiction

(<http://ausrivas.canberra.edu.au/Bioassessment/Macroinvertebrates/>) or the incompatible records discarded. Substratum cover descriptions (e.g. percent of gravel) entered as a range (e.g. “50-75”) were replaced by the range mid value. Values of some variables also suggest different methods of measurement may have been used. For example, pH values recorded at SA sites were excluded as they fell outside the range of pH readings for non SA sites. The sampling method used to identify and count macroinvertebrate taxa also had a large effect on the numbers of taxa reported at a site (Figure 8.4). Numbers of taxa reported from “lab sorts” were significantly different from all other methods and were, therefore, omitted.

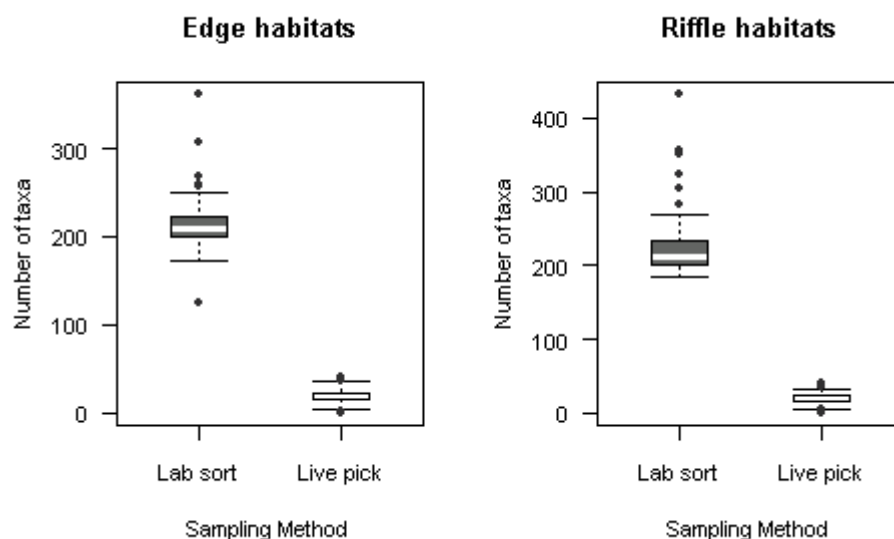


Figure 8.4 Number of taxa recorded for edge and riffle habitats at AusRivAS sites with sampling method. Macroinvertebrates are either picked and counted in the field (“Live pick”) or preserved and taken back to the laboratory for sorting and identification under a microscope (“Lab sort”).

Table 8.1 Site variables extracted from the AusRivAS database.

Variable	Variable Code
Site descriptors measured at each visit	
Average stream width (m)	AvgWidth
Turbidity (NTU)	Turbidity
Water temperature (°C)	WaterTemp
Total carbonates (mg/l)	Alkalinity
pH ₁	Ph
Conductivity (µS/cm)	Conductivity
Dissolved reactive phosphorous (µg P/l)	DR.FRP
Dissolved Oxygen (mg/L)	DO
Percent pool habitat in reach	Pool
Percent riffle habitat in reach	Riffle
Percent run habitat in reach	Run
Riparian cover descriptors measured at each visit	
Width of the riparian zone (m)	WidthOfRiparianZone
Cover of riparian zone by trees greater than 10m in height (%)	CoverByTrees.GT.10m
Cover of riparian zone by trees less than 10m in height (%)	CoverByTrees.LT.10m
Cover of riparian zone by shrubs (%)	CoverByShrubs
Cover of riparian zone by grasses, ferns and sedges (%)	CoverByGrasses.Ferns.Sedges
Percentage of the river shaded at midday	VegCoverOfRiver.Shading
Site descriptors measured for each habitat type	
Mean water depth of habitat (m)	Depth
Habitat substratum bedrock (%) ₂	Bedrock
Habitat substratum boulder (%) ₂	Boulder
Habitat substratum cobble (%) ₂	Cobble
Habitat substratum pebble (%) ₂	Pebble
Habitat substratum gravel (%) ₂	Gravel
Habitat substratum sand (%) ₂	Sand
Habitat substratum silt (%) ₂	Silt
Habitat substratum clay (%) ₂	Clay
Habitat cover by all macrophytes (%)	Macro
Habitat cover by detritus (%) ₂	Detritus
Number of Taxa ₃	NumTaxa

1. SAsites omitted
2. Values entered as ranges (e.g 50-75) replaced by range mid-value
3. Records reported with sampling method as “lab sort” omitted

8.1.1.3. Northern Territory River Habitat Survey

A survey of rivers in the Northern Territory was undertaken after attending the International Association for Landscape Ecology conference in Darwin in July 2003. Twenty-two sites (Table 8.2) were selected along a transect that followed a north-south climatic gradient in both annual average rainfall and temperature (Figure 8.5). Ideally, survey sites would be stratified according to the river environment classifications, but at the time these were not yet available. For ease of access, sites were located upstream of fords or bridges on public roads or on walking tracks in national parks or nature reserves. The length of a sampling site was the lesser of 10 times the estimated bankfull channel width and 200m, measured parallel to the stream.

Survey procedures and data sheets loosely followed the AusRivAS physical habitat assessment protocol, designed to complement the AusRivAS biological assessment of river condition (Parsons *et al.*, 2002) (Appendix 4), but with limited time and resources it was not possible nor appropriate to measure all of the variables recommended. Measurements in large rivers were particularly problematic without a boat for access (and often with crocodiles common) and thus a number of channel measurements such as cross-sectional depths were not attempted. Flow stage dependent variables in the habitat assessment protocol also presented difficulties. In particular, estimation of the extent of bedform features (e.g. rapid, riffle, pool etc.) and bars was not undertaken at sites with no flow. Other elements of the AusRivAS habitat assessment, including the Habitat Assessment method adopted from the USEPA, were also judged to be of little relevance to many Northern Territory rivers and were not applied. The USEPA method scores various habitat parameters according to categories of perceived natural or disturbed states (Appendix 4) (Parsons *et al.*, 2002). The differentiating criteria, however, have not yet been verified for Australian arid zone rivers.

Values recorded on the data sheets were entered into a Microsoft Excel spreadsheet for graphical analysis. Not enough sites were surveyed to enable tests of statistical significance. Nevertheless, exploratory analysis (boxplots, contingency tables and summary statistics) assisted appreciation of the variability among river environment groups. The NSW Rivers Survey and the AusRivAS River Health Survey enable more comprehensive testing.

Table 8.2 Northern Territory river habitat survey sites. All sites were visited between the 1st and the 19th of July, 2003.

Site Id	Name	Site description	Location		Altitude (m)
1	Ferguson River	Upstream of Stuart Highway	14° 4.1'S	131 ° 58.7'E	113
2	Edith River	Downstream of falls, campground	14 ° 10.9'S	132 ° 11.2'E	121
3	Roper River	Roper Bar (upstream of road crossing)	14 ° 42.8'S	134 ° 30.4'E	24
4	Towns River	Upstream of Roper Highway crossing	15 ° 2.7'S	135 ° 12.7'E	14
5	Limmen Blight River	Upstream of Roper Highway crossing	15 ° 28.9'S	135 ° 23.9'E	6
6	Playford River	Upstream of Tablelands Highway crossing	19° 17.4'S	136° 3.6'E	213
7	Todd River	Central Alice Springs	23 ° 41.9'S	133 ° 53.1'E	574
8	Hugh River	Upstream of Stuart Highway crossing	24 ° 21.2'S	133 ° 26.1'E	464
9	Finke River	Upstream of Stuart Highway crossing	24 ° 33.2'S	133 ° 14.3'E	412
10	Unnamed Creek	Valley of the Winds, Kata Tjuta National Park	25 ° 17.2'S	130 ° 44.4'E	589
11	Kathleen Springs	Downstream of waterhole Watarrka National Park	24 ° 20.1'S	131 ° 40.8'E	640
12	Finke River	Upstream Larapinta Drive, near Hermansburg	23 ° 57.0'S	132 ° 46.4'E	559
13	Kings Creek	Kings Canyon, Watarrka National Park	24 ° 15.0'S	131 ° 34.7'E	635
14	Newcastle Creek	Newcastle Waters, west of Stuart Highway crossing	17 ° 16.0'S	133 ° 27.1'E	250
15	Palm Creek	Finke River Gorge National Park	24 ° 2.8'S	132 ° 42.6'E	584
16	Roper River	Upstream of road crossing , Bitter Springs	14° 54.3'S	133° 5.0 E	125
17	Katherine River	1st gorge, Nitmiluk National Park	14 ° 18.7'S	132 ° 25.5'E	120
18	Douglas River	Upstream of weir @ gauging station 814063	13 ° 47.8'S	131 ° 20.4'E	55
19	Daly River	Luke's farm camp site #3	13 ° 58.9'S	131 ° 13.1'E	37
20	Mary River	Upstream boat ramp, Arnhem Highway	12 ° 54.5'S	131 ° 38.8'E	19
21	East Alligator River	Upstream of Cahill's Crossing	12 ° 25.5'S	132 ° 57.9'E	6
22	Barramundi Creek	Marguk plunge pool, Kakadu National Park	13 ° 19.0'S	132 ° 26.3'E	75

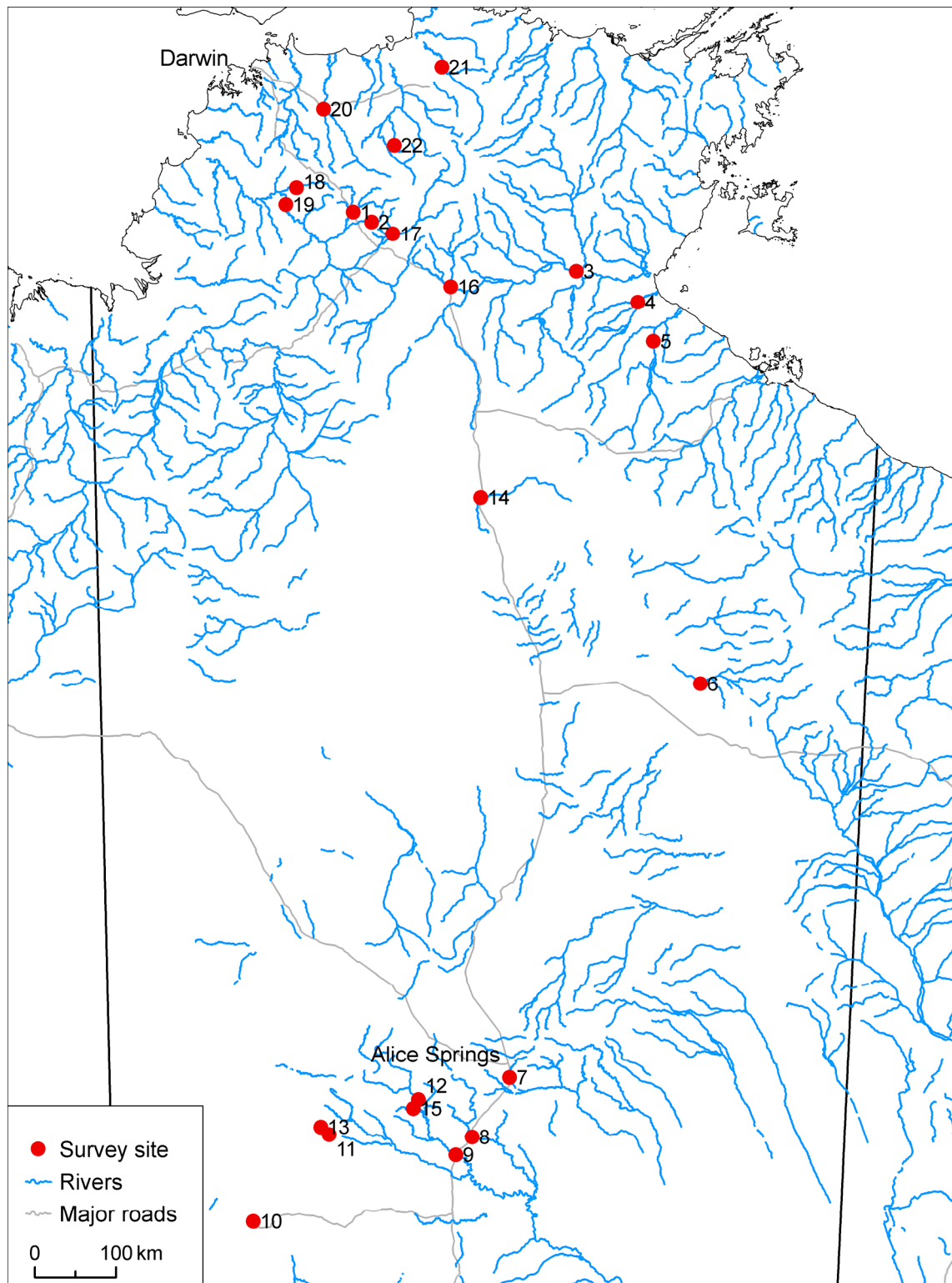


Figure 8.5 Location of Northern Territory survey sites.

8.1.2. Linking classifications and survey sites

Site locations in each test dataset were imported into Arc/Info for assignment to a stream segment by overlay onto the segment sub-catchment grid. The assigned segment identifier links the site to a lookup table containing the group affinity vectors for each river environment classification. As noted previously (Section 4.2.3), river sites may not overlay the same river's

representation in the DEM-derived stream network because the river was generalised in the DEM, its location was derived from different base maps or, especially near confluences, the grid cell in which the site is located is associated with a tributary. Routines were written to compare site information such as waterway name, catchment area, elevation and distance from source with the equivalent parameters independently attributed to the gridded segments from the DEM or the GEODATA TOPO 250K streamline mapping. A poor match identified a site that may have been allocated to the wrong stream segment.

All sites in the two smaller data sets (NSW Rivers Survey and NT river habitat survey) were manually checked against digital 1:100,000 topographic mapsheets and relocated as necessary to the closest segment grid cell, guided by the accompanying site descriptions. Two sites in the NSW Rivers Survey could not be matched adequately and were discarded. This checking was not feasible for the large number of AusRivAS sites so potentially misallocated sites were omitted from the analysis of site habitat variables. AusRivAS sites were assumed to be misallocated if the DEM-derived values of either elevation or the distance to source at the segment pour-point (the maximum flow path length) were different by more than 20% from those recorded at the site. Previous experience has found this figure to reasonably indicate mismatches. The differences were expressed relative to those of the reference value to account for the inherently larger errors in the DEM at higher elevations (Hutchinson *et al.*, 2001). Similarly, the distance to source measurements are likely to be less certain for larger catchments. Unfortunately, catchment area and stream name were not consistently attributed in the AusRivAS database and could not be used for checking. Just 580 of the 1543 AusRivAS sites attributed as “Reference” satisfied the matching criteria.

The IBRA regionalisation and sub-regionalisation version 5.1 (Environment Australia, 2000) were also evaluated using the same test data to permit comparison with the river environment classifications developed in this study. IBRA 5.1 was supplied as polygon coverages in Arc/Info export format by the Australian Department of Environment and Heritage. IBRA region numbers were assigned to sites by overlaying the sites on the IBRA polygons in Arc/Info using the IDENTITY command.

8.1.3. Evaluating ecological relevance

8.1.3.1. Biotic similarity

A modified measure of classification strength (CS) was earlier used to compare classifications with different numbers of groups (Section 7.2.3.2). Here it is used in its original formulation to measure the biotic similarity of the river environment classifications. CS is calculated as the mean of all pairwise within group similarities ($\overline{W}$) relative to the mean of all pairwise between

group similarities ($\bar{B}$) (Van Sickle, 1997; Van Sickle and Hughes, 2000). Within group similarities are weighted by the number of objects within the group and only defined for groups with 2 or more members. CS may be expressed as the difference ($\bar{W} - \bar{B}$) or the unitless ratio, ($\bar{B}/\bar{W}$). A large difference or a small ratio implies objects in the same group are substantially more similar to each other than they are to objects in other groups and hence is indicative of a strong classification (Van Sickle and Hughes, 2000).

The ASIM program in PATN version 3.6 computes the ratio $\bar{B}/\bar{W}$ using the ANOSIM algorithm attributed to Clarke and Green (Belbin, 1993b). ASIM reads the input association matrix as dissimilarity values, scaled between zero and one. Other workers (a series of papers in a special issue of the Journal of the North American Benthological Society introduced by Hawkins and Norris, 2000; Snelder, Cattaneo *et al.*, 2004) have reported the classification strength of landscape classifications for freshwater biota based on between and within group similarities, presented in the form of a mean similarity dendrogram (Van Sickle, 1997). For compatibility with these studies, the ASIM program was modified to convert the input dissimilarities to similarities (1- dissimilarity) and to write out the within group similarities for individual groups for presentation as mean similarity dendrograms. Mean similarity dendrograms were produced using a routine written in the R statistical software scripting language (R Development Core Team, 2004). This routine plots the dendrogram node at $\bar{B}$ on the similarity axis with branches for each group of length W_i where W_i is the within group mean similarity for group i . The mean similarity dendrogram of a strong classification will have relatively long branches (Van Sickle and Hughes, 2000).

Classification strength was computed for each of the river environment classifications and the IBRA regionalisation based on the compositional similarity of the fish fauna at sites from the NSW River Survey and the macroinvertebrate populations at AusRivAS reference sites. A dissimilarity matrix was computed from the site by species data matrices with the ASO program in PATN using the Bray-Curtis coefficient (Belbin, 1993b). The Bray-Curtis coefficient is equivalent to the Czekanowski coefficient for presence/absence data (Belbin, 1993b). It is recommended for ecological data (Faith *et al.*, 1987) and was adopted for comparable international studies (Hawkins and Norris, 2000; Snelder, Cattaneo *et al.*, 2004). When expressed as a similarity between objects, the Bray-Curtis coefficient takes values between zero (nothing in common) and one (perfect match).

The classification strength derived directly from *a posteriori* classification of the faunal sites suggests an upper bound to that achievable by any landscape classification for the same number of groups (Van Sickle and Hughes, 2000). Snelder and colleagues (2004) expressed the classification strength of an *a priori* landscape classification (the River Environment Classification, Snelder and Biggs, 2002) as a percentage of the CS of *a posteriori*, spatially

neutral classifications of the biotic data as an objective means of comparing CS across different test data. For comparability, this approach was followed here. CS values were computed for classifications of the site by species matrices from the NSW River Survey and AusRivAS reference sites. Classifications were generated with the flexible beta UPGMA clustering algorithm ($\beta = -0.1$) (Belbin *et al.*, 1992) as implemented in the FUSE program in PATN (Belbin, 1993b). The number of groups in these *a posteriori* classifications was set to the number of landscape classification groups represented by at least two sites, that is, the number of groups included in the calculation of $\overline{W}$.

The importance of drainage divides in structuring biological populations is widely acknowledged (Tait *et al.*, 2003). To assess whether the landscape classifications are any better at discriminating patterns in the distribution of freshwater biota, CS values were compared to those achieved by grouping sites simply according to drainage basin. A test of the null hypothesis of “no class structure” was also conducted using ASIM to randomFly permute objects between groups and compare the randomized ratio ($\overline{B}/\overline{W}$) to that of the *a priori* classification. The test was applied by determining the number of times the ratio derived for the random grouping was less than that derived for the *a priori* classifications from the default 100 permutations.

8.1.3.2. Mixed effects models

The ability of a classification to discriminate significant differences in ecologically important habitat characteristics was tested using linear mixed effects models (Pinheiro and Bates, 2000) to predict AusRivAS site variables. Through the specification of fixed and random effects, mixed effects models account for known structure in the data, such as in the spatial and temporal variability at a site contained in the repeated measurements (habitats or seasons).

Models were fitted with the LME function from the NLME library in the R statistical package (R Development Core Team, 2004) using the maximum likelihood method with fixed effects for classification group, season and / or habitat as appropriate (Table 8.1) and a random effect for site. The maximum likelihood method allows comparison of models with different fixed effects structures (e.g. alternative classifications) (Pinheiro and Bates, 2000). The significance of the classification group term was tested with conditional F-tests using the ANOVA function in R (R Development Core Team, 2004). This test provides more realistic estimates of probability values than would a likelihood ratio test for models with different fixed effects structure (i.e. with and without classification group), especially where the number of groups and hence the number of parameters removed from the fixed effects is large (Pinheiro and Bates, 2000). The explanatory power of the alternative classifications was compared with the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC). These criteria assist model

selection (Pinheiro and Bates, 2000) and are more reliable measures of goodness of fit than the usual R-squared (John Maindonald, Statistical Computing Unit, ANU pers.comm.8/5/2001). They are calculated by the ANOVA function in R as:

$$AIC = 2 * \log\text{-likelihood} + 2 * n_{\text{par}}$$

$$BIC = 2 * \log\text{-likelihood} + \log(n) * n_{\text{par}}$$

where n_{par} is the number of parameters in the fitted model and n is the number of observations (R Development Core Team, 2004). Both criteria favour more parsimonious solutions but BIC discriminates more strongly against complex models. A smaller value of AIC or BIC indicates a better model of the data.

Regression diagnostics (plots of standardized residuals versus fitted values) were checked for violation of regression assumptions for a sample of the models. Normality was assessed with normal probability plots. Where a clear lack of normality was evident, site variables were transformed with either a logit (i.e. $\ln(1/1-p)$ for proportions), square root (for counts) or natural log transformation (for measures). For proportions of zero or one the logit is not defined so these values were adjusted by a very small amount (0.00001) up or down respectively. Even though the logit is a strong transformation (Maindonald and Braun, 2001) the normal probability plots still indicated some departure from normality for those variables expressed as proportions. Nonetheless, these variables were retained because the large number of observations made the assumption of normality less critical (Maindonald and Braun, 2001). Since undertaking this analysis the LME function in R has been replaced by LMER from the LMER package (Bates and Sarkar, 2005). The new package allows generalized linear mixed models to be fitted with an appropriate error distribution and link function and so should enable such difficulties to be avoided in future mixed modelling exercises.

8.2. Results

8.2.1. Biotic similarity

Classification strength (CS), measuring the similarity of the biota at sites within a group relative to sites in other groups, was computed from the site by species matrices from the NSW Rivers Fish Survey and the AusRivAS National River Health Program.

8.2.1.1. *Fish community composition*

A total of 39 native (non-estuarine) fish species were recorded on the 97 sites used from the NSW Fish Survey. No native species were caught at two sites (sites 21.1 on the Belubula River

and 96.0 on Tarcutta Creek) while just one or two species were recorded at another 21 sites. The four most species rich sites each have 16 native freshwater species.

The CS ratio ($\bar{B}/\bar{W}$) computed for 100 random permutations of the objects between groups was always higher than that for all *a priori* landscape classifications, so the null hypothesis of “no class structure” can be rejected in each case. Classification strength results produced similar rankings of the landscape classifications for fish community composition whether recorded as presence/absence or abundance (Table 8.3). Level 5 of the hierarchical river environment classification consistently produced the best results - the lowest CS ratio and the greatest absolute and relative CS difference. The ALL classification has a similar CS to that of Level 3 in the hierarchical classification, slightly worse than either the IBRA regions or sub-regions.

Classification strength tends to increase as the groups in the hierarchical classification are successively subdivided (Table 8.3) but so too does the range of values of individual mean within group similarities as shown by the length of the branches in the mean similarity dendrograms (Figure 8.6). For example, on the basis of the fish species present or absent, the Level 1 (climate) type number 8 has a mean similarity among its member sites of just 0.23 but when sub-divided according to flow class (Level 2) within group mean similarities ranged as high as 0.66 for type 82 from a low of 0.2 for type 89. The Level 5 sub-division varies even more, from 0.17 (type 81241), indicating low assemblage compositional similarity to a perfect match similarity of 1.0 (type 87241). However, as groups are increasingly divided the relative number of types sampled by more than one fish survey site also drops (Table 8.3). At least two sites are necessary to compute within group pair-wise similarities. The similarity of the fish fauna at sites within the ALL groups also varies greatly as it does between IBRA regions and sub-regions and drainage basins (Figure 8.7) but again the number of survey sites is low for many groups.

The average between group similarity $\bar{B}$, also increased with the number of groups. For the hierarchical classifications it increased slightly but steadily from 0.17 at Level 1 to 0.22 at Level 5 for the presence/absence matrix and from 0.13 to 0.18 for the abundance matrix. There was very little difference in the $\bar{B}$ values between the IBRA regions and sub-regions (0.19 to 0.2 and 0.15 to 0.16 for the presence/absence and abundance matrices respectively).

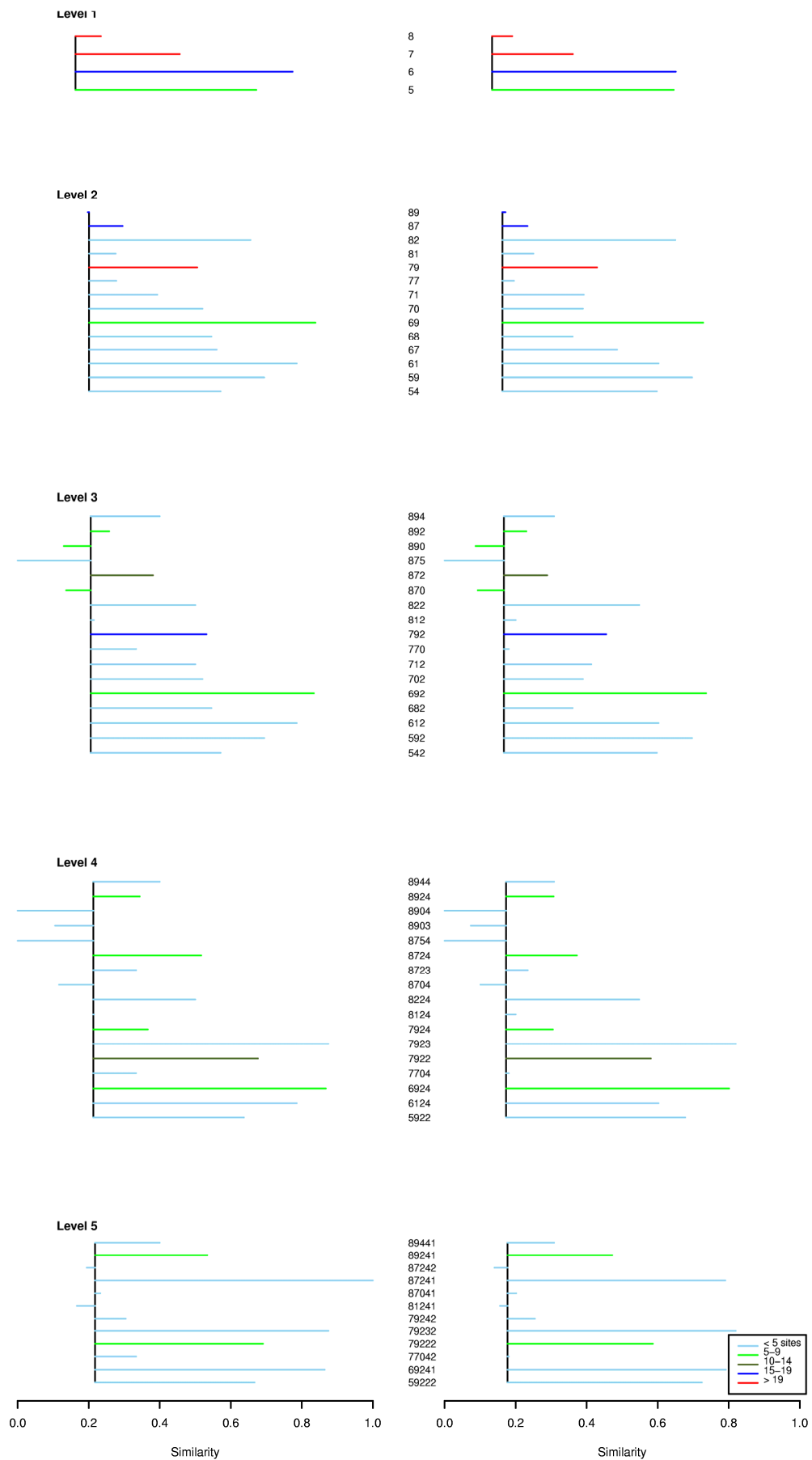


Figure 8.6 Mean similarity dendrograms for fish species presence/absence (left) and average catch (right) for the hierarchical river environment classifications. The number of sites in each group are indicated by the colour coding.

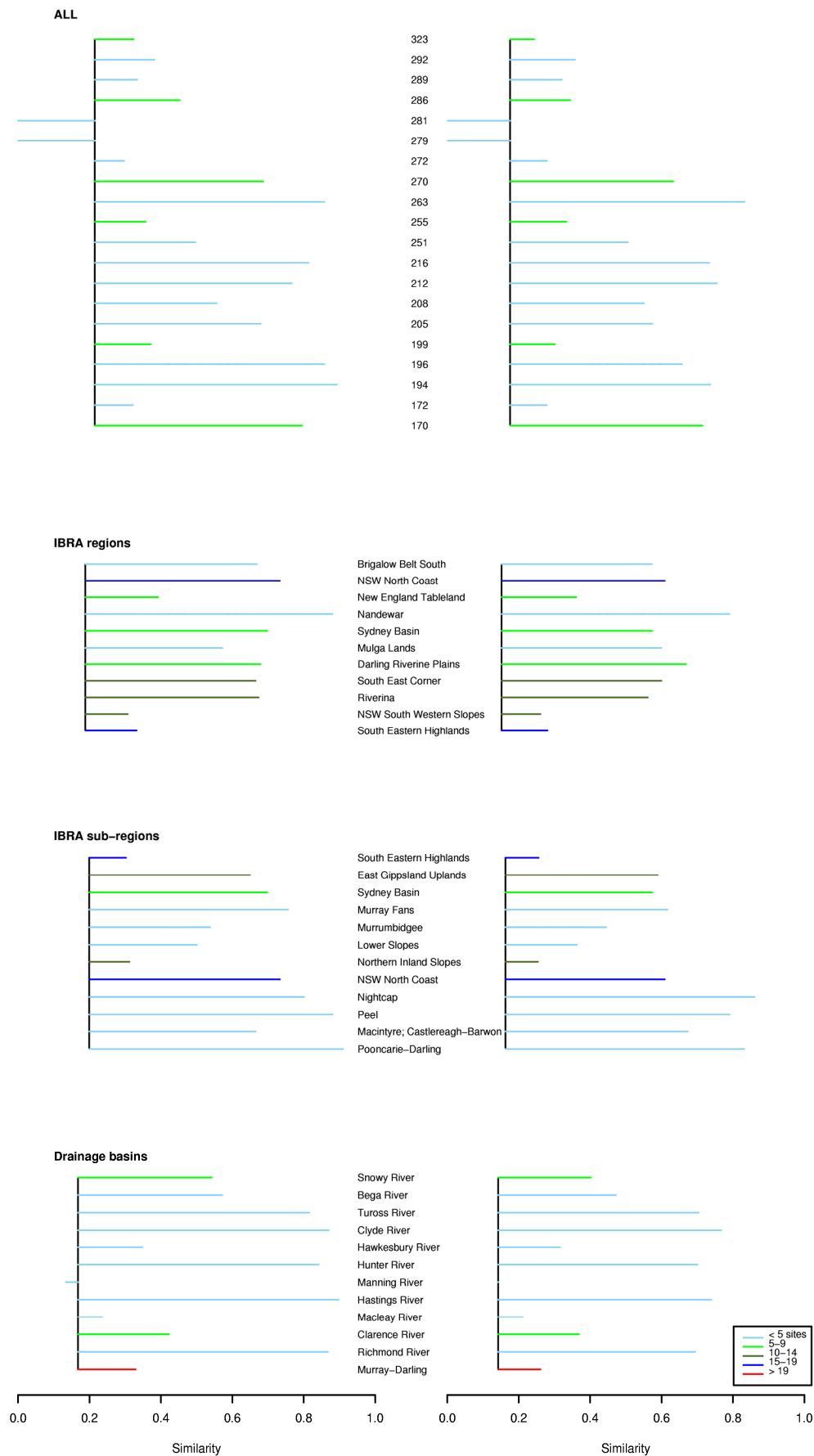


Figure 8.7 Mean similarity dendrograms for fish species presence/absence (left) and average catch (right) for the ALL stream landscape classification, IBRA regions and sub-regions and topographically defined drainage basins. The number of sites in each group are indicated by the colour coding.

Table 8.3 Classification strength of landscape classifications evaluated using fish community composition at sites from the NSW Rivers Survey database. Relative CS values are the absolute CS difference expressed as a percentage of the absolute CS difference of an *a posteriori* clustering of sites with the same number of groups as the landscape classification groups sampled by more than one site.

Classification	Total no of groups	No. of groups sampled > 1 site	Presence/absence			Abundance		
			$\bar{B}/\bar{W}$	$\bar{W} - \bar{B}$	Relative CS (%)	$\bar{B}/\bar{W}$	$\bar{W} - \bar{B}$	Relative CS (%)
Level 1	10	4	0.46	0.19	42	0.46	0.16	40
Level 2	81	14	0.50	0.20	32	0.48	0.18	33
Level 3	420	17	0.43	0.27	41	0.41	0.24	42
Level 4	2,361	17	0.39	0.34	51	0.37	0.29	52
Level 5	8,614	12	0.35	0.40	66	0.34	0.35	67
ALL	355	20	0.44	0.27	39	0.41	0.25	43
IBRA regions	85	11	0.37	0.33	55	0.34	0.29	57
IBRA subregions	354	12	0.41	0.29	47	0.39	0.25	48
Drainage basins	49,688	12	0.50	0.17	28	0.53	0.12	24

8.2.1.2. Macroinvertebrate community composition

Macroinvertebrate counts were extracted from the AusRivAS National River Health database for 1407 reference sites for a total of 204 taxa. No data were provided for another 136 reference sites in the database. The number of taxa recorded at a site ranged from 7 to 79 with a mean of 37. Even with many sites excluded because their reference status was not recorded, the AusRivAS sites represent many more of the river environment groups than does the NSW Fish Survey (Table 8.4). For both macroinvertebrate taxa occurrence and abundance, the IBRA sub-regional classification is the strongest, just slightly better than the IBRA regions and the Level 5 River Environment Types but there is very little difference between any of the landscape classifications. While the null hypothesis of “no class structure” was rejected for all river environment classifications the absolute CS values indicate that all are reasonably weak classifiers of macroinvertebrate community composition (Table 8.4). However, the performance of the landscape classifications is much better when considered relative to the CS values of *a posteriori* classifications of the sites (Table 8.4) and is similar to the relative CS observed for the fish faunal composition (cf. Table 8.3). These *a posteriori* classifications indicate an upper bound for CS (Section 8.1.3.1). Drainage basins again provided the weakest classification for the aquatic biota.

Table 8.4 Classification strength of landscape classifications evaluated using macroinvertebrate community composition at sites from the AusRivAS database. Relative CS values are the absolute CS difference expressed as a percentage of the absolute CS difference of an *a posteriori* clustering of sites of the same number of groups as there are landscape classification groups sampled by more than one site.

Classification	Total no. of groups	No. groups sampled > 1 site	Presence/absence			Abundance		
			$\bar{B}/\bar{W}$	$\bar{W} - \bar{B}$	Relative CS (%)	$\bar{B}/\bar{W}$	$\bar{W} - \bar{B}$	Relative CS (%)
Level 1	10	10	0.80	0.11	58	0.78	0.11	57
Level 2	81	56	0.80	0.12	46	0.76	0.12	45
Level 3	420	116	0.79	0.12	44	0.76	0.12	42
Level 4	2,361	207	0.77	0.14	46	0.73	0.14	47
Level 5	8,614	234	0.76	0.15	48	0.72	0.15	47
ALL	355	163	0.77	0.14	47	0.73	0.14	48
IBRA regions	85	40	0.75	0.15	62	0.71	0.15	63
IBRA subregions	354	132	0.74	0.16	57	0.70	0.16	56
Drainage basins	49,688	142	0.88	0.06	21	0.84	0.07	24

The mean similarity dendrograms for macroinvertebrate taxa presence/ absence and average counts are very similar so only the former are presented (Figure 8.8 and Figure 8.9). Like the fish similarity dendrograms, the length of the dendrogram arms become progressively more variable through the levels of the hierarchical classification (Figure 8.8). At Level 1 the dendrogram arms are consistently short – between +0.04 and +0.25 similarity units or less. In contrast, at Level 5 the length of the dendrogram arms varies between –0.37 to +0.38 similarity units. Average sample sizes also decline through the levels of the hierarchical classification. Despite the large overall number of sites in the AusRivAS dataset, it samples very few of the Level 5 River Environment Types at five or more sites.

Average within group similarities for IBRA regions are consistently better than average between group similarities (Figure 8.9). Although overall the strongest classification, some IBRA sub-regions nonetheless identify groupings of sites with very little biotic similarity (Figure 8.9). The similarity of macroinvertebrate communities among sites in the ALL groups is even more variable. For 12 of the ALL groups the average similarity between sites within the group is less than the average of the between group similarities, $\bar{B}$. Notably, $\bar{B}$ varies little across the classifications despite the large disparity in the number of groups. For example, $\bar{B}$ increases only very slightly, albeit steadily, with the fairly rapid rise in the number of groups from Level 1 to Level 5 (0.4468 at Level 1 to 0.4627 at Level 5).

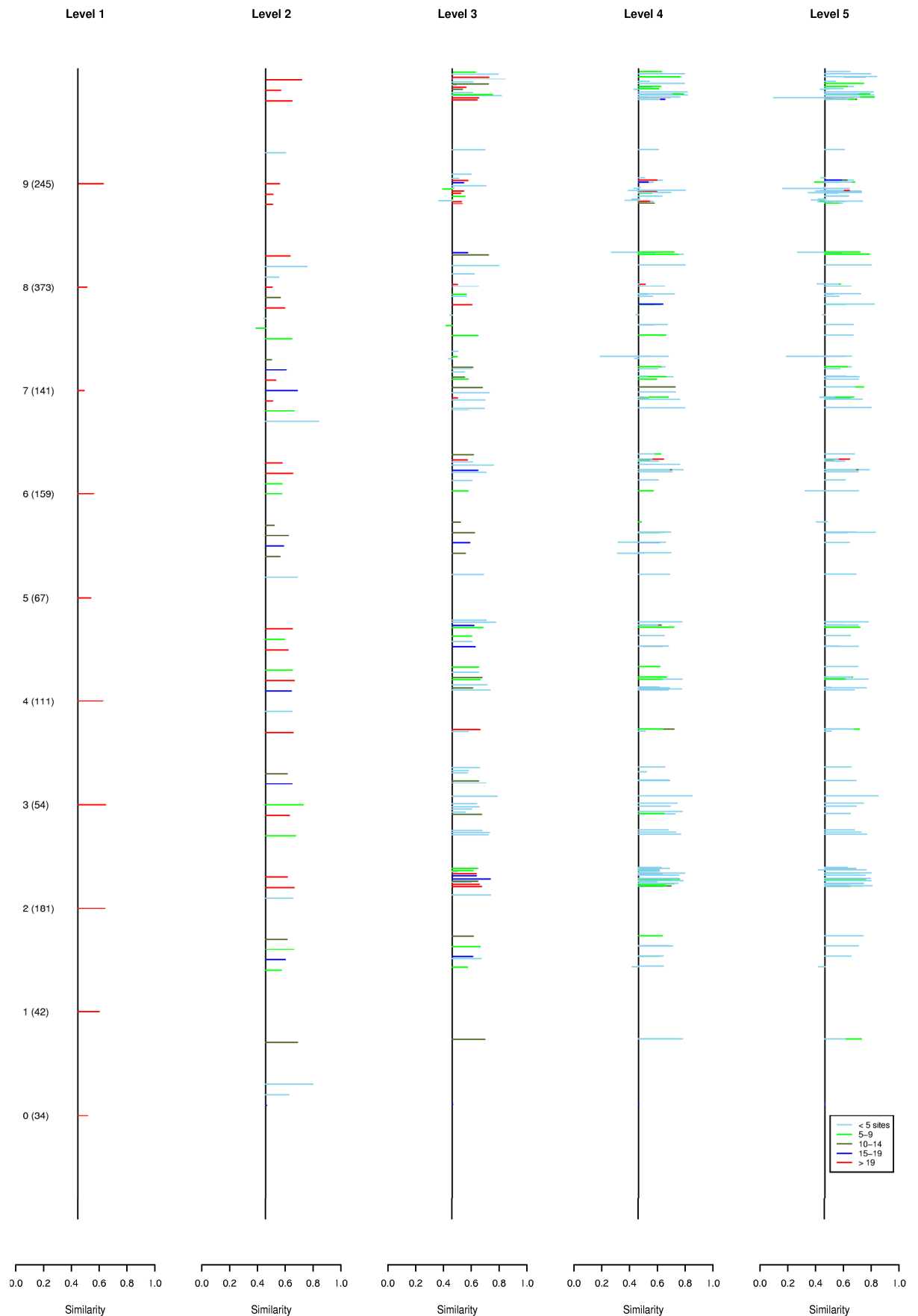


Figure 8.8 Mean similarity dendrograms for macroinvertebrate taxa presence/absence for the hierarchical river environment classifications. The numbers of sites in each group are indicated in parentheses for Level 1 types and by the colour coding for Levels 2 to 5 types.

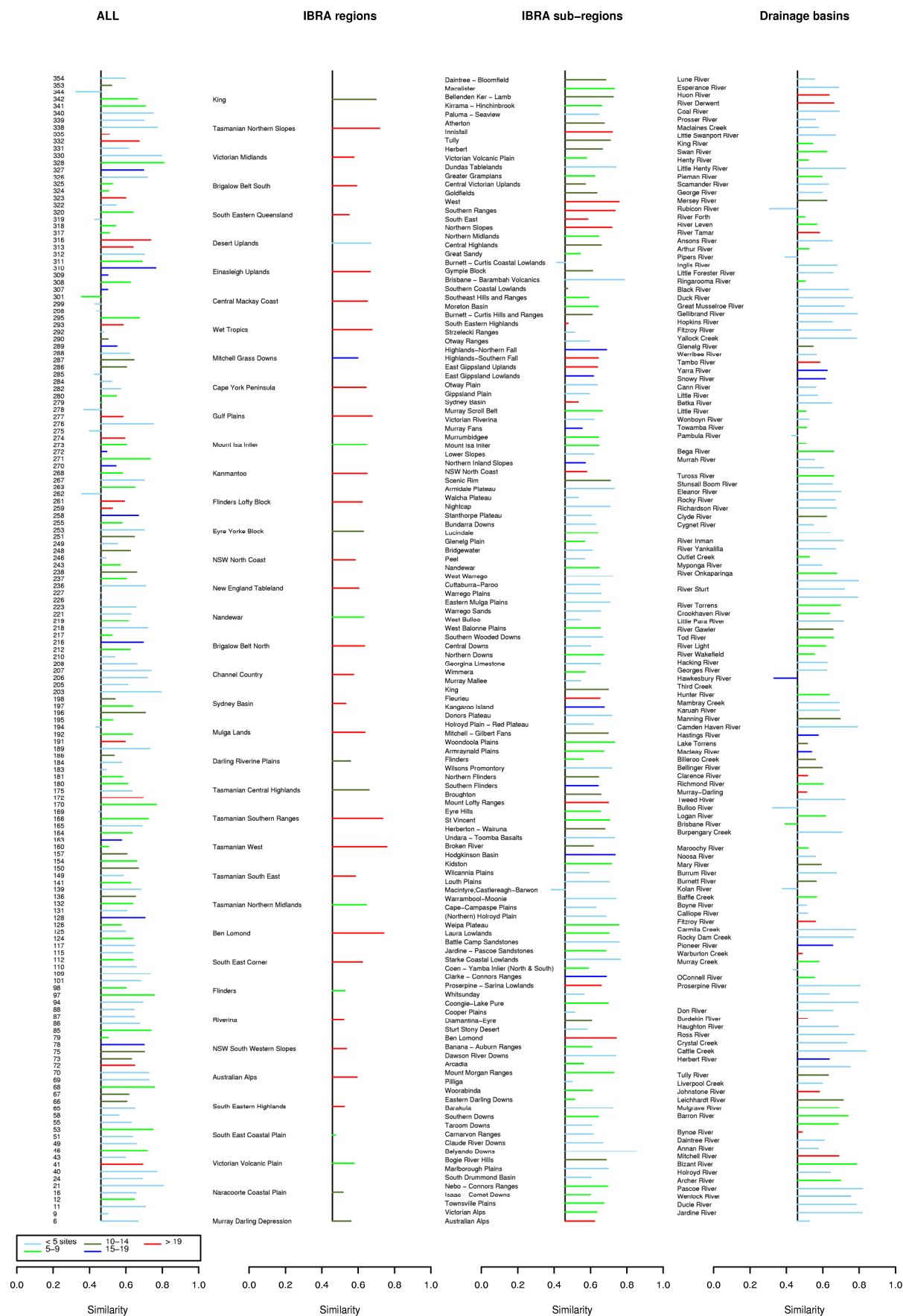


Figure 8.9 Mean similarity dendrograms for macroinvertebrate taxa presence/absence for the ALL river environment classification, IBRA regions and sub-regions and topographically defined drainage basins. The numbers of sites in each group are indicated by the colour coding.

8.2.2.Stream habitat descriptors

Across all AusRivAS site variables, the conditional F-tests strongly reject the null hypothesis that the mixed models with a fixed effect term for classification group fit the data no better than the reduced model with a fixed effect for season and / or habitat alone. The classification group term is always significant at a probability of 5% or better (Table 8.6). However, while there are a large number of observations of each variable, many are repeated measurements at the same site for different habitats and / or in different seasons or years (Table 8.6). Consequently, they represent a much smaller number of sites and hence proportion of the classification groups. Moreover, the sites are not evenly distributed among the classification groups (see Figure 8.8 and Figure 8.9). While the overall variability attributed to the classifications was significant, this was not true for all individual group coefficients in the models.

The information criteria ranked the explanatory power of the alternative classifications quite differently. As expected, BIC (Figure 8.10) strongly favoured the models with fewer parameters, ranking the model with the 10 group Level 1 classification as providing the best fit for nearly three-quarters of the site variable models and almost always including either the Level 2 or IBRA regions models in the top three places (Table 8.5). In contrast, AIC (Figure 8.11) most often ranked the models with IBRA subregions as best. Five other classifications produced models with the lowest AIC values and all classifications were positioned in the best three model fits for one or more variables.

Table 8.5 Number of variables for which the landscape classification is judged to provide the best fit or is ranked among the best three models according to the Akaike Information Criterion (AIC) or Bayesian Information Criterion (BIC).

Classification	AIC best	BIC best	AIC best 3	BIC best 3
Level 1	2	22	7	28
Level 2	0	2	6	29
Level 3	0	0	2	0
Level 4	1	0	2	0
Level 5	1	0	2	0
ALL	3	0	18	0
IBRA	4	5	25	29
IBRA subregions	18	0	25	1

Table 8.6 Summary of mixed effect models for AusRivAS site variables. Shown are the conditional F- test statistics computed for the fixed effect term, classification group. The probability of the test statistic is less than 0.001 unless otherwise indicated (*=<0.05,=0.01). Variable descriptions are given in Table 8.1. The number of observations of each variable and the number of sites at which they were recorded are shown in parentheses.**

Site variable (no.observations/no.sites)	ALL	Hierarchical level					IBRA	IBRA sub
		1	2	3	4	5		
Alkalinity (2391/673)	6.4	19.6	6.4	4.9	3.6	3.2	10.4	7.3
AvgWidth (2587/708)	10.0	9.7	19.0	9.9	6.9	7.2	8.6	5.0
Conductivity (2601/709)	10.8	32.6	12.2	7.8	7.1	7.6	30.4	16.1
DO (1823/514)	2.5	16.4	4.3	2.3	8.7	6.5	5.0	1.9
DRP.FRP (795/150)	17.3	54.2	30.9	17.5	11.0	10.7	28.8	28.5
Ph (2110/645)	4.5	25.9	7.5	4.6	3.4	4.2	10.1	5.8
Pool (519/174)	5.0	7.6	4.5	3.1	3.2	3.1	5.5	4.0
Riffle (1003/238)	4.4	14.8	6.0	4.0	3.8	3.1	8.6	5.4
Run (519/174)	3.4	6.3	3.8	2.8	2.8	2.7	4.4	3.7
Turbidity (2384/705)	8.1	23.6	11.6	7.2	5.6	5.8	17.0	10.8
WaterTemp (2573/709)	25.6	212.2	56.4	29.2	16.7	16.0	70.0	33.8
VegCoverOfRiver.Shading (1267/377)	9.3	24.4	12.2	10.1	9.3	7.5	29.5	20.3
CoverByGrasses.Ferns.Sedges (612/271)	3.6	6.2	3.3	2.5	2.4	2.4	3.9	3.6
CoverByShrubs (600/271)	2.3	9.2	3.3	2.9	2.9	2.9	4.4	3.2
CoverByTrees.GT.10m (495/232)	2.1	3.3	1.6*	1.4*	1.8	2.9	3.5	3.1
CoverByTrees.LT.10m (605/270)	2.2	4.8	2.1	1.8**	2.5	4.5	3.8	3.6
WidthOfRiparianZone (598/272)	1.9	4.1	2.2	2.2	2.2	2.8	4.9	3.9
Bedrock (4509/626)	2.3	2.1*	1.7**	1.6	1.8	2.6	3.5	2.1
Boulder (4674/643)	4.6	21.1	5.3	3.6	3.3	3.6	9.0	5.2
Clay (2916/418)	4.1	10.3	3.9	3.1	3.1	3.3	7.9	7.7
Cobble (4741/650)	6.2	23.6	7.0	5.0	4.6	4.5	12.0	7.2
Depth (4194/621)	22.1	130.7	40.9	21.4	13.8	12.0	63.0	43.6
Detritus (4174/588)	2.1	5.3	1.9	1.7	1.7	2.2	2.7	2.3
Gravel (4700/649)	3.6	12.3	4.8	3.1	2.9	3.1	5.9	4.4
Macro (1761/259)	3.6	9.0	4.5	3.1	3.9	4.2	4.7	4.2
NumTaxa (2648/305)	2.4	2.4*	2.2*	2.7	2.4	2.4	2.7	3.6
Pebble (4732/652)	5.6	22.1	7.3	4.7	4.4	4.4	10.6	6.7
Sand (4736/651)	3.2	6.1	1.9	2.3	2.5	3.7	4.8	4.3
Silt (4667/651)	3.8	10.1	4.7	3.5	3.2	3.3	6.1	4.9

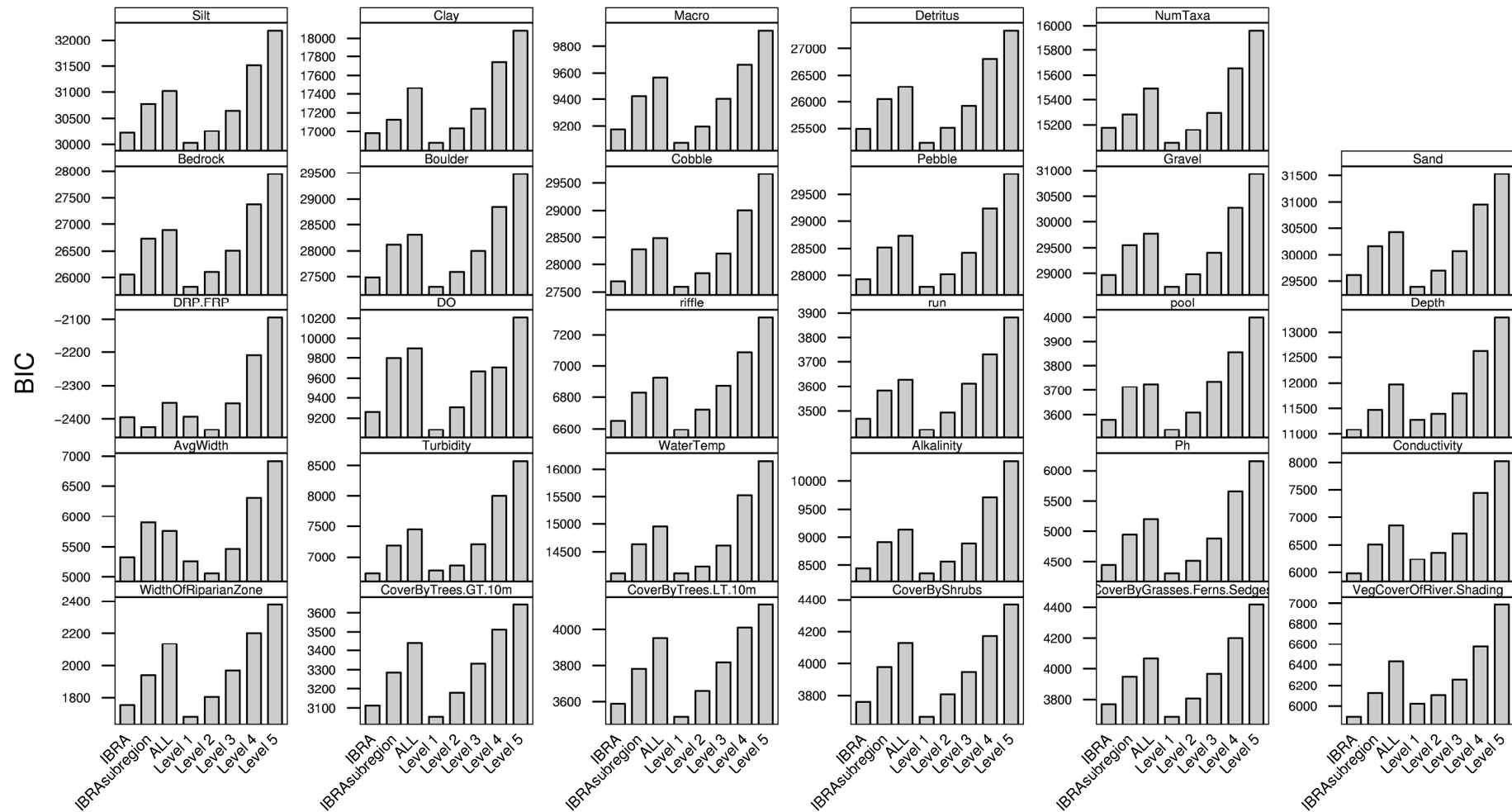


Figure 8.10 Bayesian Information Criterion (BIC) values by classification for each of the AusRivAS site variable mixed models. Lower BIC values indicate a better model of the data.

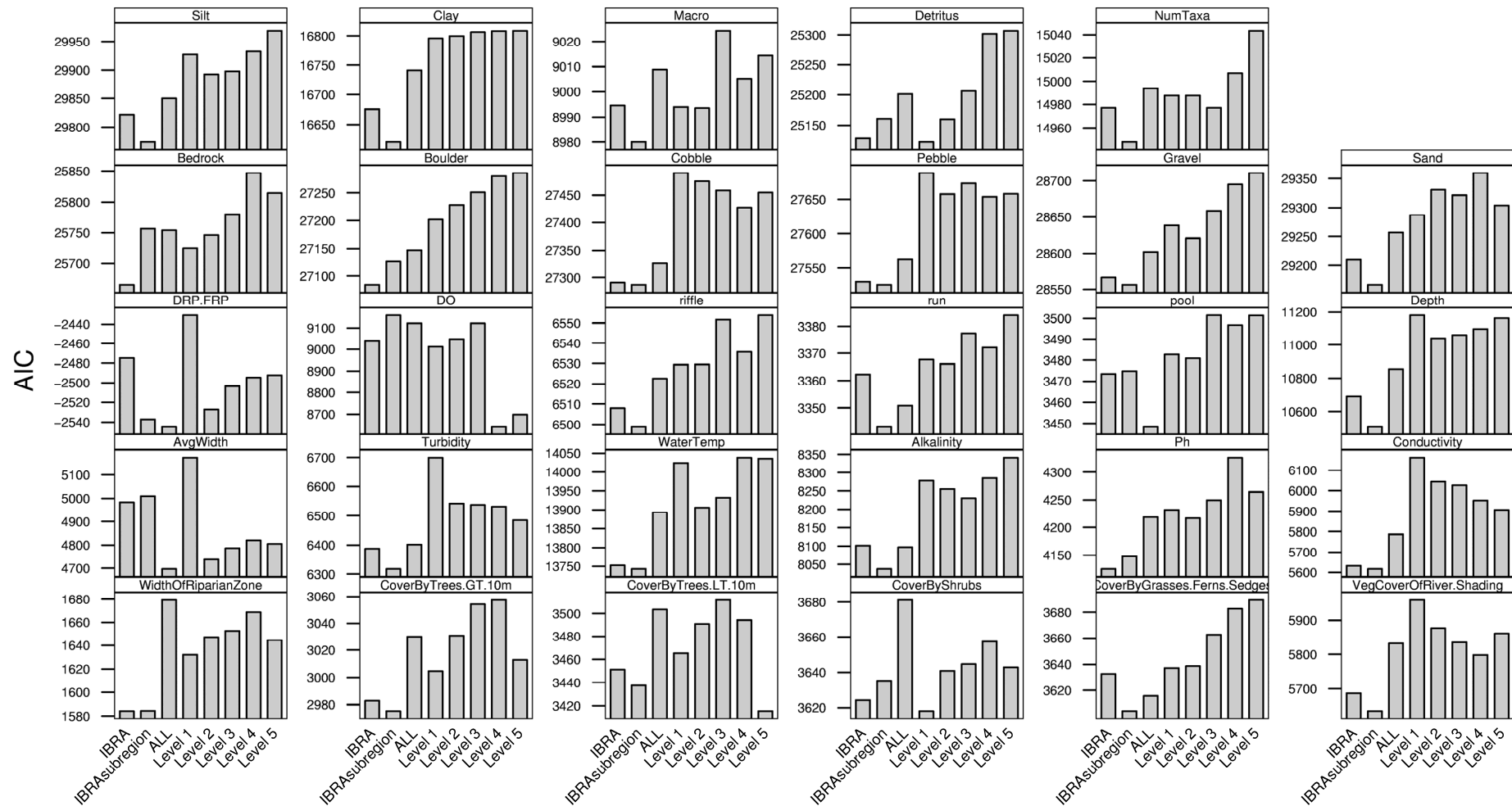


Figure 8.11 Akaike Information Criterion (AIC) values by classification for each of the AusRivAS site variable mixed models. Lower AIC values indicate a better model of the data.

8.2.3. Northern Territory river habitat survey

The river habitat survey conducted in the Northern Territory sampled a diversity of river forms (Figure 8.12). The sites surveyed ranged from small streams of just 2m wide at bankfull to large rivers up to 130m wide. Accordingly, the length of the sampling site (the lesser of 10 times the bankfull width and 200m) varied from 20 to 200m (average 168m). Sites were largely undisturbed. Nature conservation and/or recreation were the dominant landuses. More than half of the sites were located within protected areas. Although evidence of cattle grazing was seen at eight sites, only one, heavily grazed on both sides of the river, received a “very high” disturbance rating (Figure 8.13). Nineteen of the sites fell into the “low” or “very low” categories for overall vegetation disturbance. Site 7, on the Todd River flowing through the centre of the township of Alice Springs (Figure 8.12), was assigned a disturbance rating of “high” using the criteria in the habitat assessment protocol (Parsons *et al.*, 2002) (Appendix 4). Although vegetation was generally undisturbed at the sites selected, many had evidence of human activities likely to produce local impacts downstream. More than half, for example, were upstream of a road crossing, either a bridge or a ford. Litter was noted at seven sites.

Graphical exploration of the data suggested that the landscape classifications do distinguish important variation in habitat characteristics among the surveyed rivers, but the small number of observations precluded tests of the significance of these differences. Figure 8.14, for example, shows the distribution of values for selected site variables for the Level 2 River Environment Types. The structural characteristics of the riparian vegetation clearly differ between the Level 2 types. Sites of types 6 and 36 recorded the highest riparian cover by trees greater than 10m in height, but relatively low levels of shading of the stream by vegetation. This is because these were also the widest streams with bank full widths of greater than 60m. The proportion of cover by grasses, ferns and sedges were most clearly discriminated by the Level 1 groupings with little overlap among the four Level 1 groups represented (Figure 8.14). The floristic composition of the riparian zone also varies with classification group. The riparian vegetation at 10 of 11 sites on rivers in Level 1 types 0 and 3 (climate groups 1 and 4, Table 7.10) was dominated by *Melaleuca* species. The remaining site in these groups was located in a closed canopy forest. River Red Gums, *Eucalyptus camaldulensis*, fringed every surveyed stream in the Level 1 type 5 (climate group 6).



1. Ferguson River
36222



2. Edith River
31242



3. Roper River
06222



4. Towns River
01222



5. Limmen Blight River
06223



6. Playford River
14226



7. Todd River
53045



8. Hugh River
53235



9. Finke River
54226



10. Unnamed Creek, Kata
Tjuta NP 53256



11. Kathleen Springs
32206



12. Finke River
54226



13. Kings Creek
53256



14. Newcastle Creek
16243



15. Palm Creek
53246



16. Roper River
06222



17. Katherine River
06442



18. Douglas River
31222



19. Daly River
36222



20. Mary River
36223



21. East Alligator River
36247



22. Barramundi Creek
31242

Figure 8.12 Northern Territory survey sites with Level 5 River Environment Type



Figure 8.13 Cattle grazing, Playford River

The extent of bars was quite variable within some Level 2 types, though there could be substantial error in some observations. The extent of bars is ‘the percentage of the streambed area that protrudes to form a bar of any type’ (Parsons *et al.*, 2002, p.128) but was difficult to estimate reliably for dry streams or braided systems with poorly defined channels. Considerable variation in bed and bank substrates between sites was also observed (Figure 8.14). The dominant particle size of the streambed varied from fine silts to bedrock though most often was sand. The particle size distribution was better discriminated by the classifications incorporating additional landscape factors (e.g. Level 4 or the ALL classification) but very few of these types sampled more than one site making it impossible to draw any inference about the variability of the stream bed within a type.

Likewise, too few sites were surveyed to draw any conclusions on the distribution of habitat features within classification groups, but some patterns are suggested from analysis of contingency tables. In many cases, only one of the possible categories was observed for sites in a group. For instance, vegetated side bars were exclusively present or absent from sites in the Level 5 groups, although more than one site occurred in only 5 of the 17 groups. Similarly, a distinct floodplain was solely present or absent at sites in most of the ALL classification groups (Table 8.7)

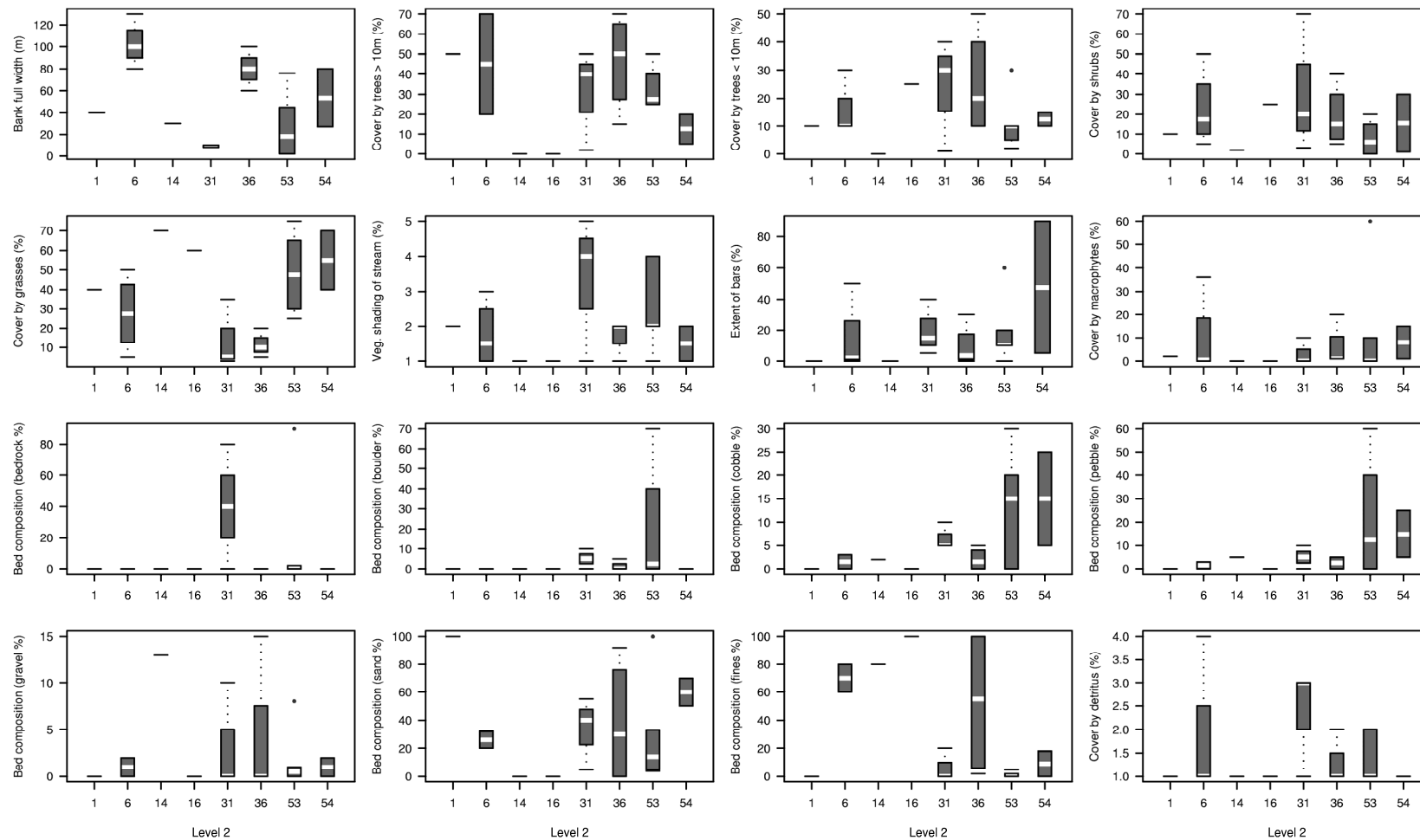


Figure 8.14 Boxplots for selected attributes with Level 2 River Environment Types for Northern Territory survey sites. The Level 2 type labels are formed from the combination of the climate and flow classification group numbers (Section 7.3.2).

Table 8.7 Number of Northern Territory river survey sites by floodplain occurrence with ALL classification group.

Character	ALL group												
	5	11	16	28	29	33	55	60	87	98	133	158	171
Floodplain absent		1				2	1	1		1		4	
Floodplain present	1		1	1	2	1			1		3		1

8.3. Discussion

The tests of ecological relevance confirm that the landscape classification groups are indeed ecologically meaningful. The fish and the macroinvertebrate assemblages found in streams within a group are more similar than they are to streams in other groups. A range of instream and riparian habitat descriptors also differ significantly between groups.

However, the landscape attributes used in classifying stream segments structure a range of potential ecosystem states in both space and time. Smaller scale ecosystems develop within the constraints imposed by these large scale landscape factors (Frissell *et al.*, 1986), dependent on processes operating over much shorter time scales (Naiman *et al.*, 1992). The available test data capture a snapshot of dynamic ecosystems. Consequently, landscape classifications should not be expected to encapsulate all of the variation in assemblage composition or habitat features measured by the sample data.

Even so, the best of the river environment classifications achieve CS values about two-thirds of those of direct *a posteriori* classifications of the biotic communities. Interpretation of relative CS values warrants some caution, however. The CS results calculated for *a posteriori* classifications of the site biota are considered to represent the maximum achievable by a classification with the same number of groups, but this number of groups did not necessarily reflect the level of environmental partitioning. For instance, the Level 5 classification divides stream segments into more than 8500 groups nationwide, but the NSW Rivers Fish Survey samples just 12 of these groups at more than one location. Consequently, relative CS values are based on a *a posteriori* classification of the fish data constrained to 12 groups.

Tests with a single indicator are not an appropriate basis to evaluate the ability of a classification to represent ecosystem variation (Omernik, 1995). Accordingly, both biotic and abiotic ecosystem components were considered in this study, albeit independently. Habitat variables were tested individually, though it would have been preferable to combine them in a way that depicted their aggregate influence on the provision of stream habitat. Developing an

aggregate measure that also accounts for the spatio-temporal variability at a site contained within the AusRivAS repeat measurements is a challenge for the future.

8.3.1. Comparing classifications

No classification performed substantially better than every other across all tests of ecological relevance. The Level 5 classification was clearly strongest when judged by the similarity of the fish community, but was only slightly weaker than the IBRA regionalizations for macroinvertebrate assemblages. For about the same level of partitioning of environmental space the combined classification (ALL), performed comparably with the hierarchical Level 3 or Level 4 classifications (for fish and macroinvertebrates, respectively) and marginally weaker than the IBRA subregions.

Classification strength was qualitatively consistent across both measures of assemblage similarity (presence/absence or abundance) and both forms of CS (the ratio, $(\overline{B}/\overline{W})$ and the difference, $(\overline{W} - \overline{B})$) permitting some confidence in the reliability of conclusions concerning the relative strengths of different classifications. CS values generally indicated a stronger classification for presence/absence data, consistent with the expectation of greater similarity for less informative assemblage descriptors (Van Sickle and Hughes, 2000).

Drainage divides act as significant barriers to movement of obligate freshwater taxa (Unmack, 2001) so greater biotic similarity within basins might be expected. However, drainage basins provided a relatively weak classification, notably so for the macroinvertebrates. The results for macroinvertebrate similarity are not unexpected, however, as the AusRivAS surveys include taxa with terrestrial life stages and hence greater ability to distribute across drainage divides (Tait *et al.*, 2003). CS results confirm that spatial units finer than drainage basins are required to adequately represent aquatic biodiversity.

It is difficult to draw conclusions as to which classification is the strongest from the site variables mixed models. The two model selection criteria, AIC and BIC, produce very different rankings. Both criteria penalise models with a large number of parameters and so favour the classifications with fewer groups, excessively in the case of BIC. The most parsimonious solution is not always that which is most ecologically meaningful. The selection of the 10 group Level 1 (climate) classification so often by BIC is not consistent with knowledge of stream processes. The grain size distribution of the channel substratum, for example, is related to catchment geology and stream power, the latter a function of discharge and slope (Knighton, 1998). As both factors will vary greatly among streams within a Level 1 group it was expected that classifications that account for their influence (i.e. Level 4 or ALL) would better explain the AusRivAS substratum grainsize variables. AIC discriminates less strongly against

classifications with larger numbers of groups and therefore selects a greater number of the classifications, though most often favouring the IBRA sub-regionalisation.

8.3.2. Comparison with other studies

The absolute values of the CS differences for the River Environment Types in this study are considerably higher than the low values reported for ecoregions from comparable international studies (Table 8.8). They are also greater, although less markedly so, than those for New Zealand's River Environment Classification. The latter is a particularly pleasing result as the REC classification was mapped at a spatial scale of 1:50,000 (Snelder and Biggs, 2002), much finer than was possible for this study. Such comparisons should really take into account the maximum achievable CS. Unfortunately, relative CS values were not always reported by these international studies. Furthermore, as noted above (Section 8.3.1), the relative CS values computed in this study may not be reliable because of the large number of groups not sampled.

In contrast to a couple of the international studies (Gerritsen *et al.*, 2000; Waite *et al.*, 2000), but in common with others (Hawkins and Vinson, 2000; McCormick *et al.*, 2000; Snelder *et al.*, 2005), the average of the between group similarities ($\bar{B}$) varied little for a given data set, regardless of the number of groups in the classification. This implies that the increased similarity between some groups that accompanies the finer partitioning of stream segments is of a similar magnitude to the greater dissimilarity between others. Low variation in $\bar{B}$ will also arise when overlap in assemblage composition between groups constrains the range of possible values of between group similarities. This is the case for both test data sets. One subfamily of macroinvertebrates, the Tanypodinae (non-biting midges), was recorded at 1338 (95%) of the 1407 AusRivAS reference sites. Another ten taxa were present at more than three-quarters of the sites. Similarly, of 39 species of native fish recorded by the NSW Rivers Survey one species, the Australian Smelt (*Retropinna semoni*), occurs on 64 of the 97 sites while another four: Flathead Gudgeon (*Philypnodon grandiceps*), Golden Perch (*Macquaria ambigua*), Gudgeon (*Hypseleotris spp*) and Long-finned eel (*Anguilla reinhardtii*) were recorded from about a third or more of the sites.

Notably, the IBRA regionalisations performed better than ecoregions elsewhere (Table 8.8) and comparably with the drainage network intelligent, river environment classifications. The performance of the sub-regionalisation was generally better than that of the broader IBRA regions. Concerns have been raised about the suitability of IBRA for aquatic biota (Wells and Newall, 1997; Turak *et al.*, 1999; Marchant *et al.*, 2000) but the subregions have not been previously tested for aquatic systems (Tait *et al.*, 2003).

Table 8.8 Summary of classification strength results for landscape classifications from international studies adopting comparable methods. Compositional similarity was measured by the Bray-Curtis or Czekanowski coefficients for abundance and presence/absence data respectively. Absolute CS values are the best CS ($\overline{W} - \overline{B}$) results reported for either presence/absence or abundance similarity matrices. Taxon: A=amphibians; F=fish; I=invertebrates

Reference	Location	Classification	Number of groups	Number of test sites	Taxon	Absolute CS
Snelder, Cattaneo <i>et al.</i> , 2004	New Zealand	REC Level 1	3	237	I	0.06
		REC Level 2	5			0.09
		REC Level 3	11			0.11
		REC Level 4	15			0.11
Van Sickle and Hughes, 2000	Western Oregon, USA	Ecoregion	4	137	A+F	0.13
McCormick <i>et al.</i> , 2000	Mid-Appalachian Highlands, USA	Ecoregion	9	200	F	0.03
Waite <i>et al.</i> , 2000	Mid- Atlantic Highlands, USA	Ecoregion	5	88	I	0.01
Sandin and Johnson, 2000	Sweden	Ecoregion	6	428	I	0.06
Feminella, 2000	Southeastern USA	Ecoregion	4	30	I	0.13
Gerritsen <i>et al.</i> , 2000	Wyoming, USA	Ecoregion	4	85	I	0.09
Hawkins and Vinson, 2000	Western USA	Ecoregion	24	1912	I	0.07

These results are not, however, conclusive evidence of the classification strength of IBRA. The effects of spatial autocorrelation can confound evaluation of the power of regionalisations to distinguish variation among streams because the similarity of sites increases simply with proximity (Magnusson, 2004). Thus, even where regional boundaries confer no ecological meaning, ecological descriptors will be more alike within a region if the spatial autocorrelation length is long relative to the size of a region (Wolock *et al.*, 2005). Comparison of CS values with suitable distance based neutral-model classifications is required to provide stronger evidence for the strength of the IBRA regionalisations relative to other non-geographic, river environment classifications (Van Sickle and Hughes, 2000). Such neutral models would account only for the distance between sites and could be generated simply by clustering sites according to between site geographic distances or allocating them to randomly located regions derived at multiple scales (Van Sickle and Hughes, 2000).

8.3.3. Variability in the strength of individual groups

Global tests of classification strength and the significance of the classification term in the mixed models indicate that there are important differences among at least some of the groups. The river environment classifications always partition sites better than a random placement into groups. There is, nonetheless, considerable variability among groups in both a) within group similarity (W) and b) the significance of parameters in the mixed models. This implies that the groups are not equivalent in their ability to represent the variation in riverine ecosystem structure. Such variability has also been noted by other studies (Hawkins *et al.*, 2000).

There are many possible explanations. In addition to uncertainty contributed by the landscape classifications (Section 7.4.3.2), some variability is likely to be due to the limitations of the test data. The distribution of both the NSW Fish Survey and AusRivAS sites among the River Environment Types was very uneven with many of the river types not sampled by either survey. The proportion of groups with few sites (< 30) increases with the number of divisions in the classification. Thus, some of the observed variability of W with increasingly finer partitioning of streams is likely to be an artefact of random sampling effects associated with small sample sizes (Hawkins and Vinson, 2000; Heino *et al.*, 2004).

Sampling effects linked to the spatial heterogeneity within a site may also have influenced test results. Macroinvertebrate communities from the different habitats contained within a site may be quite distinctive (Parsons and Norris, 1996). Counts from all habitats were aggregated to facilitate assessment of biotic similarity between sites. However, sampling effort was not proportional to the area occupied by a habitat and not all habitats present at a site were always sampled. In Victoria, for example, the main channel was not sampled in lowland rivers and only the slow flowing, “edge” habitats were sampled at all sites (Marchant *et al.*, 2000). Thus, sampling bias may also have confounded the detection of biotic differences between some classification groups.

The AusRivAS protocols require macroinvertebrates to be identified only to family level in most cases (e.g. Nichols *et al.*, 2000). Taxonomic resolution is thus another issue potentially affecting within group biotic similarity, but the significance of its effect on classification strength is unclear (Hawkins *et al.*, 2000). Family level data tended to indicate weaker landscape classifications than either species or genus level data in several studies (Hawkins and Vinson, 2000; Van Sickle and Hughes, 2000; Waite *et al.*, 2000), but comparable strength in another (Feminella, 2000).

Human activities, both instream and within the catchment, can affect the composition of biotic communities (Boulton and Brock, 1999), masking differences between groups and so confounding the evaluation of classification strength. Although intended to represent minimally impacted conditions, the criteria for selection of AusRivAS reference sites vary between jurisdictions and, of necessity, consider “best available” sites in highly modified landscapes. In NSW, for example, selection criteria can result in inclusion of sites on streams with dams in their upper reaches and altered sediment regimes from grazing in the catchment (Turak *et al.*, 1999), while most reference sites in Victoria are affected to some degree by modifications to instream or catchment processes (Barton and Metzeling, 2004). The NSW Rivers Survey selected sites randomly within broad geographic regions and consequently include rivers in a wide range of condition (Harris and Silveira, 1997). Plotting the average level of anthropogenic disturbance of stream segments in a group with W suggests the impact of human activities could be responsible for at least some of the variability of W (Figure 8.15). The highest values of W are usually found for groups at either end of the disturbance spectrum. Thus, it is hypothesised that the fish communities at sites relatively undisturbed by human activities will better reflect the influence of landscape controls, or conversely, at the other end of the spectrum have been reduced to a few ubiquitous species less sensitive to gross disturbance. The effect of intermediate levels of disturbance on biotic assemblages will vary, depending on the nature of the disturbance and the relative sensitivities of different species. Such variation might, therefore, obscure the similarity between sites within a group and contribute to their low W values.

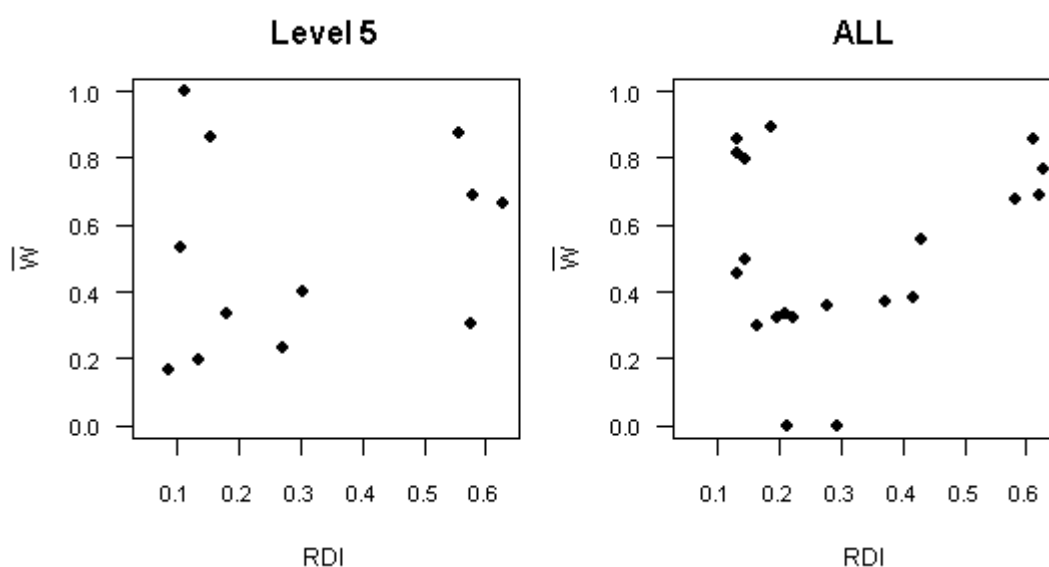


Figure 8.15 Average River Disturbance Index (RDI) values with NSW Rivers Survey presence/absence mean within group similarities (W) for Level 5 and ALL River Environment Types. The River Disturbance Index (Stein *et al.*, 2002) (Section 9.1.2) provides an indicator of the potential disturbance to natural stream processes from human activities. It is scaled along a continuum of disturbance from 0 (no disturbance) to 1 (severely disturbed).

8.4. Conclusions

The testing of the river environment classifications confirms their ecological relevance. The classifications partition the variation in stream biota and habitat characteristics among streams significantly better than would a random partition or one based on hydrological connectivity alone. Having satisfied the classification requirements of systematic conservation planning, the river environment classifications could be applied to a variety of planning tasks, including a preliminary analysis of the comprehensiveness and adequacy of the existing protected area system (Chapter 9). Applications should be consistent with the classification's scale. Scale divergence in both space and time between river sites and any landscape level classification must be considered. The classifications depict broad landscape scale patterns in river ecosystem characteristics, so they should not be expected to predict precisely the biotic composition or the physical and chemical properties of the stream at a sampled site without additional, local scale information.

Most applications require a single landscape classification so must choose between the hierarchical and ALL classifications. The decision is not clear-cut. There is no "right" classification. Chapter 7 concluded that the hierarchical classification had produced more easily interpreted and communicated River Environment Types but this was at the cost of reduced parsimony. On the tests of biotic similarity, the hierarchical classification also performs better so, on balance, might be preferred for most conservation planning applications. However, recognizing a larger number of types will influence conservation planning outcomes. For example, more rivers may be required to establish a representative system of protected rivers (Chapter 9).

The IBRA sub-regionalization performs comparably with the river environment classifications on the tests of biotic composition for macroinvertebrates, although weaker for fish. The IBRA regionalizations also rank highly among the best model fits for many of the habitat descriptors. It would be misleading, however, to use these results to support arguments that IBRA could provide an equally useful framework for conservation planning. Further testing is required to confirm that these results are not simply due to spatial autocorrelation effects. Concerns about the suitability of the model selection criteria used to evaluate the habitat predictive models (i.e. excessive penalisation of larger numbers of groups) also need to be addressed. A conceptual model of stream system function and behaviour underpins the river environment classifications. In contrast, the placement of IBRA boundaries was focussed on the needs of terrestrial ecosystems. Critical catchment constraints on the distribution of obligate freshwater biota were not considered.

The tests also show that groups within a classification may vary widely in their ability to detect ecologically meaningful differences between streams. This variability is likely due to a

combination of several factors, related to limitations in both the classifications and the test data. It further contributes to uncertainty about the relative performance of the landscape classifications. More conclusive statements concerning the relative merits of different approaches to classifying river environments will require more comprehensive test data collected from purposefully designed river surveys.

Chapter 9. Reviewing the National Reserve System

The development of a continental landscape framework to support systematic conservation planning for Australia's rivers and streams is described in parts 1 and 2 (Chapters 4 to 8). A review of existing protection measures establishes the requirements for new protective measures and hence is a vital step in systematic conservation planning (Box 1.1). Protected areas or 'reserves' are the foundation of conservation programs worldwide (Margules and Pressey, 2000). In this chapter the landscape framework is applied to a review of the protected area networks of the States and Territories that together form the National Reserve System (NRS).

This review determines the extent of streams contained within the NRS and to what degree the reserves protect streams and their biota from the impacts of upstream landuses and fragmentation. The comprehensiveness and adequacy of the NRS is assessed by evaluating the level of protection of the River Environment Types (Chapter 7), as surrogate river ecosystems. Protection levels are compared to conservation targets that take account of the natural rarity and vulnerability of the ecosystem. In conjunction with an indicator of anthropogenic disturbance, this assessment can prioritise rivers and streams for conservation action. By analysing protection levels for River Environment Types from both the 5 level hierarchical and combined (ALL) classifications the influence of classification strategy on conservation assessment is also explored.

The chapter begins with preparatory tasks, including acquisition and preparation of the protected area spatial coverage, development of the indices of anthropogenic disturbance used as indicators of vulnerability and calculation of conservation targets for River Environment Types (Section 9.1). The results of the review are presented in two parts: an overview of the extent of protected stream, and assessment of the comprehensiveness and adequacy of the NRS (Section 9.2). The chapter ends with a discussion of the implications of the results for the NRS, their limitations and suggestions for future reviews and the effects of classification choices (Section 9.3).

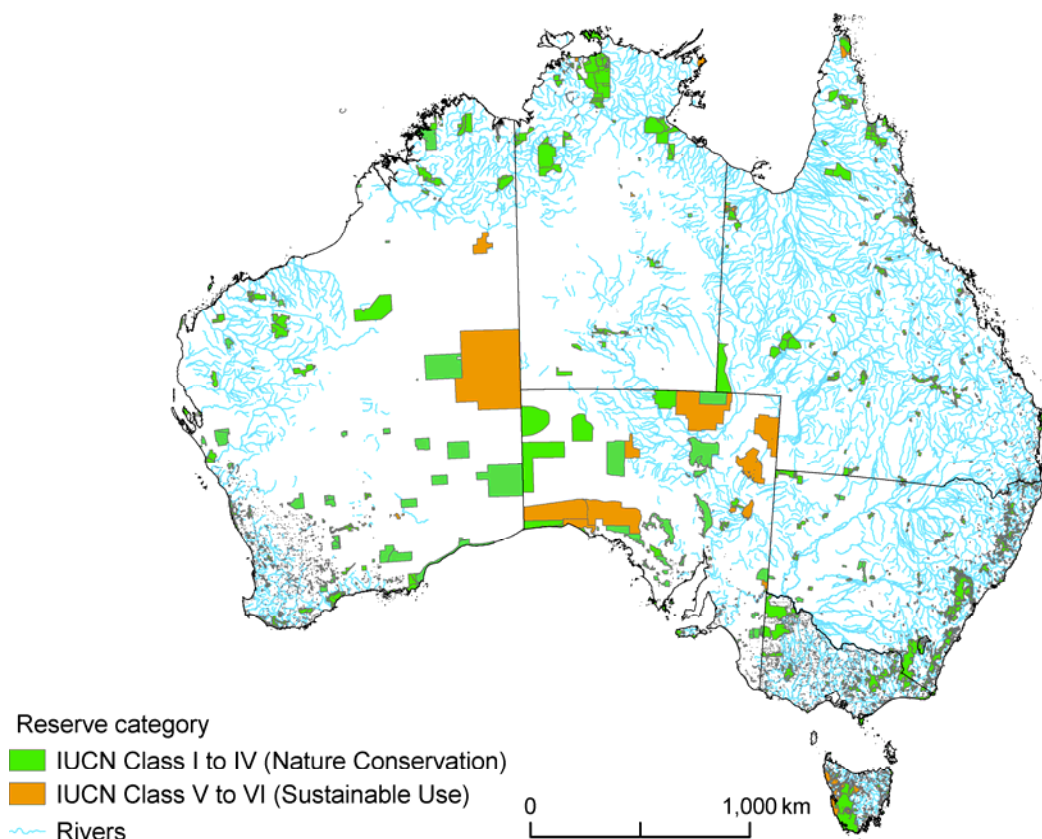


Figure 9.1 The National Reserve System (CAPAD 2002)

9.1. Methods

9.1.1. The National Reserve System

The Department of Environment and Heritage provided spatial coverage of protected area boundaries from the Collaborative Australian Protected Areas Database (CAPAD 2002). Updated regularly, CAPAD compiles spatial and textual data for protected areas declared by each of the States and Territories (Environment Australia, 2003). The data supplied includes all protected areas satisfying the IUCN definition of a protected area that had been gazetted by October 2001 with some newer areas added up until September 2002 (Figure 9.1).

The polygon coverage was gridded at 9 seconds with the Arc/Info GRID POLYGRID function to match the resolution and extent of the spatial framework (Chapter 4). Where the majority of a grid cell was overlaid by a protected area it was allocated the numeric code of the protected area's IUCN category. For analysis, this grid was also converted to a binary grid (protected area/not protected area) with zeros in cells not overlaid by an IUCN category I to IV protected area. IUCN categories I to IV include areas where the principal management intent is nature conservation but excludes areas managed primarily for their scenic or recreational values or for sustainable use (Table 9.1).

Table 9.1. IUCN guidelines for protected area management categories (after Environment Australia, 2003)

IUCN category	Description
Ia. Strict Nature Reserve: Protected Area managed mainly for science	Area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features and/or species, available primarily for scientific research and/or environmental monitoring.
Ib. Wilderness Area: Protected Area managed mainly for wilderness protection	Large area of unmodified or slightly modified land and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed so as to preserve its natural condition.
II. National Park: Protected Area managed mainly for ecosystem conservation and recreation	Natural area of land and/or sea, designated to (a) protect the ecological integrity of one or more ecosystems for this and future generations, (b) exclude exploitation or occupation inimical to the purposes of designation of the area and (c) provide a foundation for spiritual, scientific, educational, recreational and visitor opportunities, all of which must be environmentally and culturally compatible.
III. Natural Monument: Protected Area managed for conservation of specific natural features	Area containing one or more specific natural or natural/cultural feature which is of outstanding value because of its inherent rarity, representative or aesthetic qualities or cultural significance.
IV. Habitat/Species Management Area: Protected Area managed mainly for conservation through management intervention	Area of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitats and/or to meet the requirements of specific species.
V. Protected Landscape/Seascape: Protected Areas managed mainly for landscape/seascape conservation and recreation	Area of land, with coast and seas as appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, cultural and/or ecological value, and often with high biological diversity. Safeguarding the integrity of this traditional interaction is vital to the protection, maintenance and evolution of such an area.
VI. Managed Resource Protected Areas: Protected Area managed mainly for the sustainable use of natural ecosystems	Area containing predominantly unmodified natural systems, managed to ensure long-term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services to meet community needs.

9.1.2. Indices of anthropogenic disturbance

Indices of anthropogenic disturbance will be used to develop a measure of vulnerability for each River Environment Type to help set conservation targets and to determine the extent of undisturbed stream available for addition to the protected area network. The indices rank streams along a continuum from near pristine to severely disturbed according to the intensity and extent of human activities that potentially impact on stream hydrological, geomorphological and ecological processes. They were computed for each stream segment following the method of Stein *et al.* (Stein *et al.*, 1998; Stein *et al.*, 2002).

The grid-based procedure, implemented in ArcInfo (ESRI, 1996), first computes factor scores based on activities only within the stream segment and its direct sub-catchment (Figure 9.2). The factor scores account for major sources of catchment disturbance (landuses, settlements and structures, infrastructure, extractive industries and other point sources of pollution) and direct alterations to flow as indicated by instream and floodplain structures (e.g. impoundments, canals and levee banks). Scores reflect both the spatial extent and potential magnitude of impact from the disturbance and, in the case of point sources, proximity to the stream. Factor scores are weighted and combined to produce two summary indicators of local disturbance: the Sub-Catchment Disturbance Index (SCDI) and the Segment Flow Regime Disturbance Index (SFRDI). The Catchment Disturbance Index (CDI) and Flow Regime Disturbance Index (FRDI) integrate, respectively, the SCDI and SFRDI scores of all upstream stream segments, weighted by their estimated contribution to streamflow, thereby indicating the degree of anthropogenic disturbance catchment-wide. Finally, an indicator of overall disturbance, the River Disturbance Index (RDI) is computed as the average of the segment CDI and FRDI scores. All index values are standardized to range between zero (undisturbed) and one (maximum disturbance).

Application of the method produced the first national assessment of river disturbance, supporting the identification of Australia's remaining "wild" rivers (Stein *et al.*, 2001). It relied on an extensive database detailing the location and type of human activity, compiled from existing national and jurisdictional databases and a drainage analysis of a 250m resolution DEM (for details see Stein *et al.*, 1998). The DEM was developed from similar source data to the national 9 second DEM version 1, projected to Lambert Conformal Conic projection coordinates.

This study replaces the original assessment of river disturbance (<http://www.heritage.gov.au/anlr/code/arc-maps.html>) with a new database of disturbance

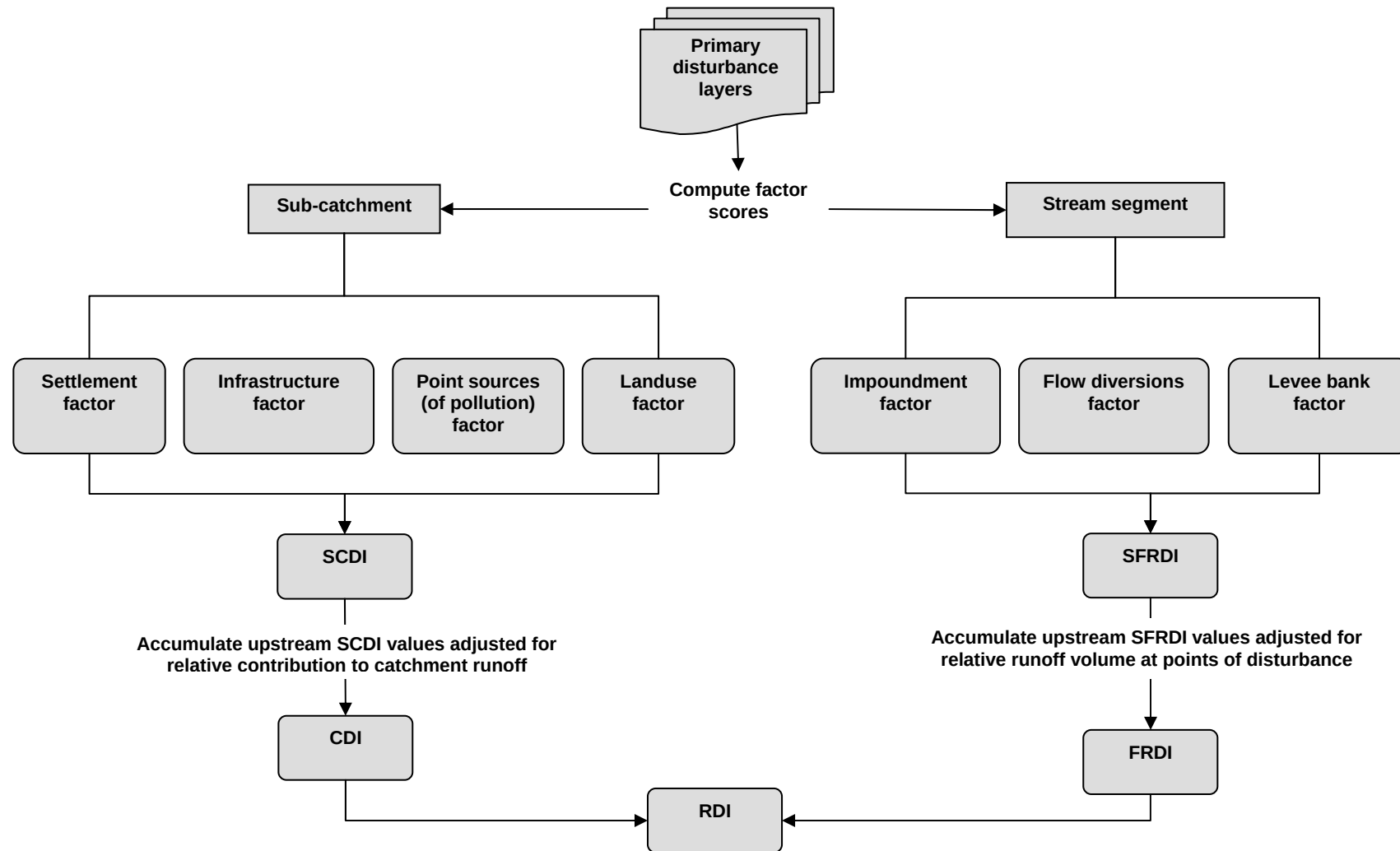


Figure 9.2 The River Disturbance Model used to derive indices of anthropogenic disturbance (after Stein *et al.*, 1998). See text for description.

indices based on the substantially upgraded DEM and utilising the products of the new drainage analysis methods (Figure 9.3). All elements of the database development dependent on the drainage analysis of the DEM were substituted with the equivalent grids from this study. The flow path distance to stream is necessary to modify the impact rating of point sources of pollution according to their proximity to the stream. This grid was derived by summing the distance to move across the surface from one cell to the next in the direction of flow until a stream is reached. Following Stein *et al.* (Stein *et al.*, 1998), contributions to total streamflow were estimated from the annual mean soil water surplus computed with the water balance module of the GROCLIM program from long-term monthly mean climate values (Hutchinson, Houlder *et al.*, 2000).

The sources of data on human activities are those described for the original assessment of river disturbance (Stein *et al.*, 1998) with two exceptions where more current disturbance information was substituted:

1. All disturbance data previously sourced from the GEODATA TOPO 250K Series 1 database (e.g. infrastructure, settlements, drainage canals) were replaced with the equivalent data from the GEODATA TOPO 250K Series 2 database (Geoscience Australia, 2003c). New features in GEODATA version 2 such as seismic lines and sewerage ponds were also coded with the relevant disturbance category.
2. The Integrated Vegetation Cover (2003), Version 1 dataset (Bureau of Rural Sciences, 2003) replaces earlier land cover data used to indicate areas of non-natural (i.e. cleared) vegetation. This data set was compiled by the Bureau of Rural Sciences from five other data sets:
 - i. Agricultural Land Cover Change (Barson *et al.*, 2000);
 - ii. Forests of Australia 2003 (National Forest Inventory, 2003) including plantation information from the National Plantation Inventory 2001 (Wood *et al.*, 2001);
 - iii. 1996/97 Land Use of Australia, Version 2 (Stewart *et al.*, 2001);
 - iv. Land Use Mapping at the Catchment Scale (Bureau of Rural Sciences, 2002) being a compilation of datasets available as of July 2003;
 - v. National Vegetation Information System 2000 (National Land and Water Resources Audit, 2001a)

The disturbance layers were converted to grids, one for each category of activity (Stein *et al.*, 1998), but unlike Stein *et al.* (1998) the data were not projected for consistency with the 9 second resolution drainage analysis products (Figure 9.3).

The Integrated Vegetation Cover dataset was supplied as a grid with a resolution of 100m in Albers projection co-ordinates. It was geometrically transformed with the ArcInfo PROJECT

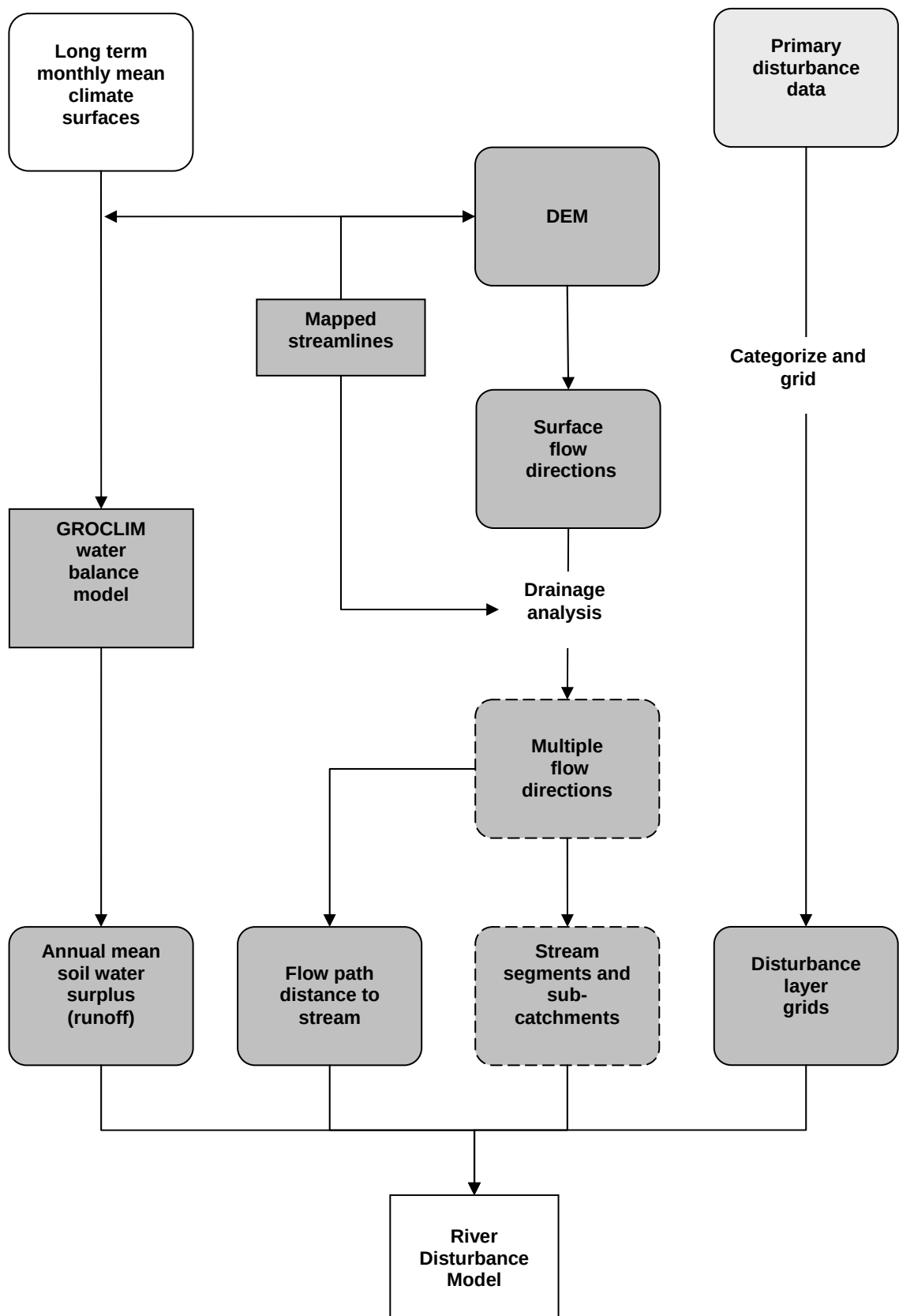


Figure 9.3 Developing the river disturbance database (modified after Stein *et al.*, 1998). Shaded boxes show database components updated by this study with dashed lines indicating products of the new drainage analysis methods. Primary disturbance layers were only partially updated (see text).

function to geographic co-ordinates with a grid cell size of 4.5 seconds (0.00125 degrees of latitude and longitude). The resampling of the grid associated with the transformation will inevitably introduce some error. For land cover categories the appropriate resampling method is nearest neighbour assignment since interpolation of values is not possible. This resampling method allocates the value of the closest cell in the original grid to the cell centre in the transformed grid (ESRI, 2004), potentially omitting rare values and distorting the distribution of values. To minimise error, the resolution of the transformed grid was chosen to be that which was closest to the original grid resolution (i.e. 100m) and was an equal divisor of the 9 second resolution employed in this study, hence 4.5 seconds. The proportion of the larger 9 second cell in a landuse category was then computed from the number of the smaller cells allocated to that category divided by 4. The river disturbance procedures, implemented in the ArcInfo AML language (ESRI, 1996) were altered to accommodate the land cover data derived from higher resolution data sources but otherwise operate largely as explained in Stein *et al.* (1998). Figure 9.4 displays the revised River Disturbance Index values used to derive a measure of vulnerability for the River Environment Types (see below).

9.1.3. Conservation Targets

As described earlier (Chapter 1) quantitative targets enable existing conservation measures to be reviewed. Conservation targets designate how much of each of the biodiversity features (here river ecosystems) should be included in a protective management framework to ensure their long-term persistence. Uniform percentage targets have been widely adopted. For example, a target of 15% of the pre-1750 distribution of each forest ecosystem is used for the Regional Forest Agreements (JANIS (Joint ANZECC / MCFFA National Forest Policy Statement Implementation Sub-committee), 1997). Similarly, a figure of 20% of river types is agreed as a conservation target for South African rivers (Jeanne Nel, CSIR, pers. comm. 20/12/05). More flexible targets (e.g. Smith *et al.*, 2002; Pressey *et al.*, 2003; Weitzell *et al.*, 2003) recognize that some biodiversity features may need greater levels of protection than others and are preferred by many conservation biologists (Pressey *et al.*, 2003).

A variable conservation target was computed for each river type using the method of Pressey and Taffs (2001) adapted for River Environment Types. Their formulation has been used to derive areal targets for land systems in western NSW (Pressey and Taffs, 2001) and for forest types in north-eastern NSW (Pressey, Watts *et al.*, 2004). It acknowledges the need for conservation action varies according to the vulnerability of the ecosystem to threatening processes, and that larger percentage targets will be necessary for rare types (Pressey, Watts *et al.*, 2004).

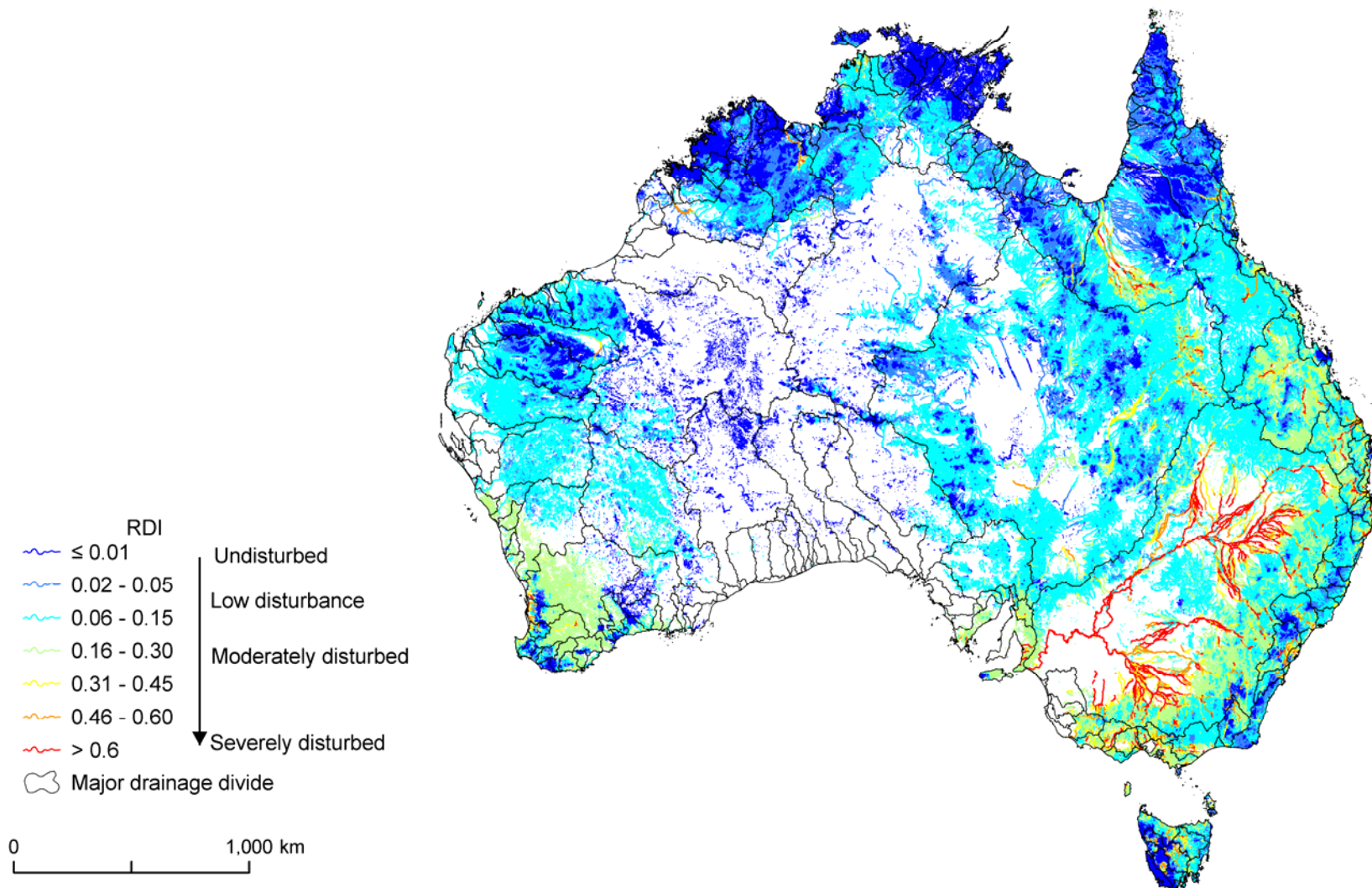


Figure 9.4 River Disturbance Index (RDI) values. RDI is used to compute a measure of vulnerability for River Environment Types.

Thus the conservation target, as a percentage of the total stream length of each River Environment Type, is defined by:

$$\text{TARGET} = 10\% + (10\% * \text{NR}) + (20\% * \text{V})$$

where NR is the natural rarity of the type and V its vulnerability. NR was computed as $(L_{\max} - L_i) / L_{\max}$ where L is the total length of stream in River Environment Type i and $L_{\max}$ is the total length of stream in the River Environment Type with the longest length. V was calculated as the average stream length weighted RDI. RDI takes account of a broader range of threats than the normalized clearing rate used by Pressey and colleagues (2004). Both NR and V are scaled between zero and one, so the conservation targets cannot be less than 10% nor exceed 40% of the total stream length.

The targets computed for River Environment Types ranged between 11 and 35%. Average percentage targets varied little between classifications, but considerably when expressed as target stream lengths in kilometres (Table 9.2). Predictably, as the number of River Environment Types increases and so the number of segments of a type decreases, the average target length of stream declines. Target lengths of less than 1km are required for hierarchical combinations that exist for just one stream segment while tens of thousands of kilometres of stream would require protection for other River Environment Types. Despite these large disparities between River Environment Types, the total length of stream required across all types generally increases consistently with the number of types defined. To satisfy the targets for the 8614 Level 5 types, for instance, the required stream length is about one third more than that for the ten Level 1 types. The total length required for the ALL classification types is relatively higher than the equivalent hierarchical classification of a similar number of types.

Table 9.2 Computed conservation targets for River Environment Types by classification

Classification		No groups	Target (%)			Target stream length (km)			
			Min.	Mean	Max.	Min.	Mean	Max.	Total
ALL		355	11.1	22.0	32.4	32	1,548	12,703	549,595
Hierarchical	Level 1	10	11.4	17.5	21.3	11,743	42,753	74,107	427,526
	Level 2	81	11.5	22.8	32.6	< 1	6,138	57,192	497,219
	Level 3	420	11.5	22.8	33.4	< 1	1,236	40,775	519,077
	Level 4	2,361	11.3	22.6	34.6	< 1	237	16,760	559,431
	Level 5	8,614	10.8	22.8	34.8	< 1	66	7,020	570,163

9.1.4. Evaluating the National Reserve System

Summary statistics, including the total length of protected stream and the areal proportion of a drainage basin that is protected, were calculated using a combination of ArcInfo functions and commands (e.g. ARC FREQUENCY, GRID ZONALSTATS) and grid algebra. The flow accumulation programs (Section 4.2.1.4) were used to compute the proportion of the upstream area of a stream that was protected. Stream segments potentially impacted by an impoundment were identified by either being upstream of a major dam (i.e. within the dam's catchment) or within the flow path downstream of a dam. The location of major dams was inferred from the values of the Impoundment Factor, one of the indices of anthropogenic disturbance (see above). A stream segment with an Impoundment Factor of 1 indicates the presence of a major dam (Stein *et al.*, 2002).

The comprehensiveness of the NRS was determined by counting the number of River Environment Types that were represented. Its adequacy was evaluated by comparing the length of protected stream of each River Environment Type to its computed conservation target and the extent of replication of types within the protected area network. Drainage divides may act as barriers to prevent the spread of disease or other catchment constrained threats to aquatic biota. Hence, including replicate samples of ecosystems from different drainage basins increases the probability of persistence of their biota. Replication was assessed by counting the number of types that were located within a protected area from more than one drainage basin. The extent of undisturbed stream outside of the existing protected area network indicates the potential to address gaps in the NRS. The additional stream length required to achieve conservation targets for each River Environment Type was computed as a proportion of the remaining unprotected and undisturbed stream length. Streams were considered to be undisturbed if their RDI score was less than 0.05 (Stein *et al.*, 2002).

The degree of protection of rare and / or threatened river ecosystems afforded by the NRS was assessed by counting how many rare and / or threatened River Environment Types were sampled by the protected area network. The status of River Environment Types was assigned following the criteria developed for forest types (JANIS (Joint ANZECC / MCFFA National Forest Policy Statement Implementation Sub-committee), 1997) and adopted by the National Reserve System program (Natural Resource Management Ministerial Council (NRMMC), 2005). Thus, a rare River Environment Type was defined as one where the total of the direct sub-catchment areas was less than 1000ha. River Environment Types reduced to <30% and <10% of their original extent by anthropogenic disturbance were labelled as threatened and endangered, respectively. A stream segment was assumed no longer representative of a River Environment Type if the overall level of disturbance measured by the RDI was greater than, or

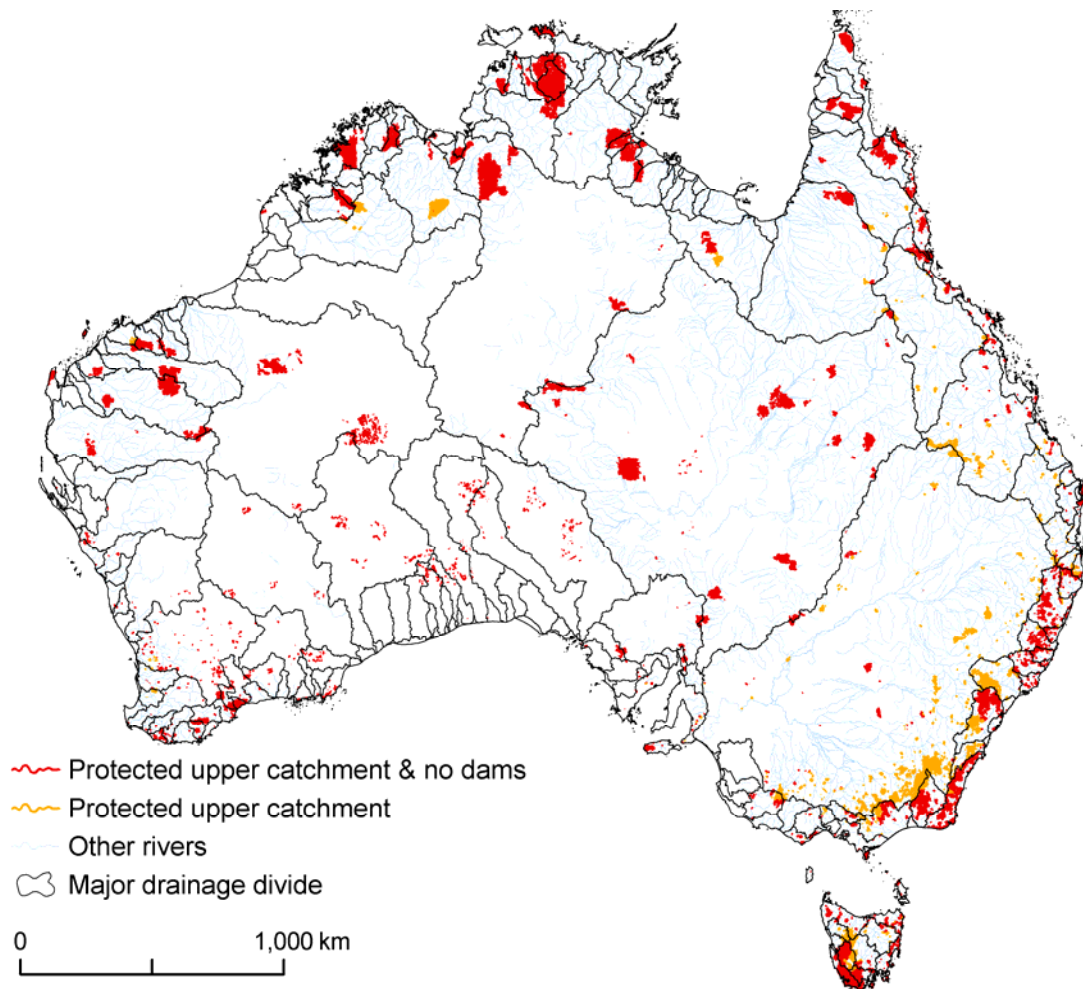


Figure 9.5 Protected rivers and streams. Sections of streams with upper catchments entirely protected.

equal to 0.3. This value indicates there is reasonably extensive disturbance throughout the catchment, intensive activities likely to have substantial localised impacts on stream character and behaviour or the presence of major instream structures associated with flow regulation.

9.2. Results

9.2.1. Protected streams and basins

Approximately 220,000km of the 2.8 million kilometres of streams fall within a protected area, 189,519km within a protected area of IUCN management classes I to IV (Table 9.3). However, activities occurring up, or downstream, of the reserved section of stream potentially threaten nearly half (108,000km) of this protected length. Of the 139,000km of stream with the entire upper catchment contained within an IUCN class I to IV protected area, about 20% has a major instream barrier within or downstream of the protected section. Less than 3,000km of named rivers have fully protected upper catchments and no major dams impeding movement of biota between their source and outlet (Figure 9.5). There are few large rivers among them – just four

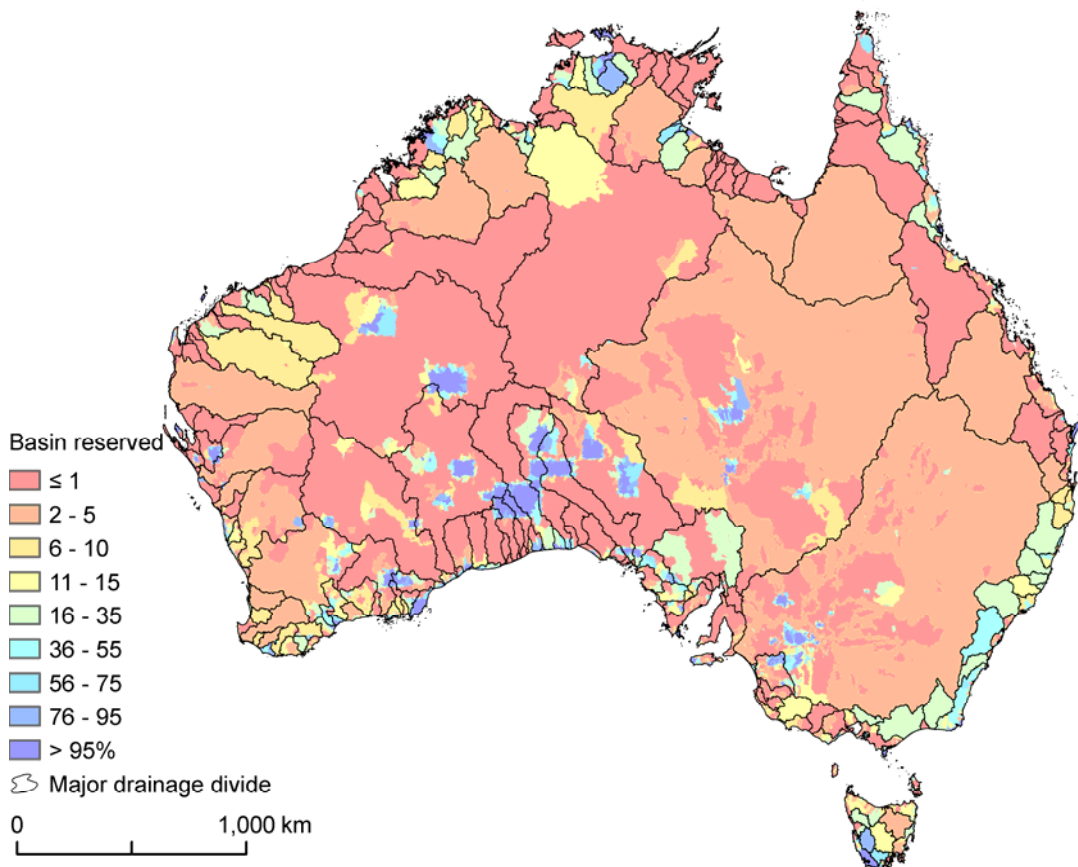


Figure 9.6 Proportion of topographically defined drainage basin contained within an IUCN category I to IV protected area.

have an estimated annual mean catchment water balance of greater than 1,000GL (Table 9.4). Two large protected areas, the Southwest National Park in Tasmania and Kakadu National Park in the Northern Territory contain many of these protected rivers.

It was not possible to identify with certainty drainage basins contained entirely within a protected area because of discrepancies in the base mapping for the drainage basin and protected areas spatial layers. In particular, where the mapping used for the protected area spatial coverage places the coastline inland of that of the drainage basin, the basin is reported as incompletely protected. Accordingly, a drainage basin was considered to be totally protected if more than 95% of its area fell within an IUCN category I to IV protected area (Figure 9.6). Just 57 of the topographically defined drainage basins with a channel network substantial enough to be named on the 1:250,000 scale topographic maps series satisfy this condition. Minor streams drain most of the substantially protected drainage basins. Only four have an area greater than 500km² (Table 9.5). The majority of the fully protected basins are small, draining arid areas to either a claypan or dry lake. A stream network is largely absent from these basins.

Table 9.3 Length of streams (km) within protected areas by IUCN category and relative size as indicated by the stream name

IUCN category	Unnamed	Creeks	Rivers	Total
Ia	16,466	1,269	626	18,361
Ib	10,538	2,130	1,282	13,950
II	125,523	15,682	12,033	153,238
III	941	217	197	1,355
IV	1,782	454	379	2,615
V	2,006	228	286	2,519
VI	22,455	2,539	1,625	26,619
Unspecified	847	102	96	1,044
Total length in a protected area	180,558	22,621	16,523	219,703
Total length all streams	2,260,953	378,922	166,018	2,805,893

Table 9.4 The 20 largest named streams (or sections of streams) with 100% of their upper catchment within an IUCN protected area class I to IV and no major dams up or downstream.

Stream name	State	Annual mean water balance (GL) ₁	Protected area names
South Alligator River	NT	1641	Kakadu National Park
Franklin River	Tas	1442	Southwest National Park
Coen River	Qld	1247	Mungkan Kandju National Park
Davey River	Tas	1056	Southwest National Park
Ray River	Tas	713	Southwest National Park
Jane River	Tas	655	Southwest National Park
West Alligator River	NT	521	Kakadu National Park
Collingwood River	Tas	446	Southwest National Park
Old River	Tas	430	Southwest National Park
Giblin River	Tas	421	Southwest National Park
Jardine River	Qld	392	Jardine River National Park
Crossing River	Tas	298	Southwest National Park
Picton River	Tas	274	Southwest National Park
Weld River	Tas	266	Southwest National Park
New River	Tas	242	Southwest National Park
Maxwell River	Tas	240	Southwest National Park
Barramundi Creek	NT	238	Kakadu National Park
Olga River	Tas	228	Southwest National Park
Hardwood River	Tas	220	Southwest National Park
Solly River	Tas	207	Southwest National Park

1. Computed to the most downstream point on the river that satisfies the criteria

Table 9.5 Fully protected drainage basins. The 20 largest drainage basins of named streams with no major dams and more than 95% of their area protected within an IUCN category I to IV reserve.

Basin name	State	Basin area (km ²)	Protected area name
Rudall River	WA	3391	Rudall River National Park
West Alligator River	NT	1375	Kakadu National Park
Spring River	Tas	1126	Southwest National Park
Davey River	Tas	838	Southwest National Park
Copper Mine Creek	WA	356	Fitzgerald River National Park
New River	Tas	301	Southwest National Park
Rocky River	SA	224	Ravine des Casoars Wilderness Protection Area
Weanerjungup Creek	WA	151	Cape Arid National Park
Saltwater Creek	Qld	109	Cape Melville National Park
Mawuwu Creek	NT	107	Garig Gunak Barlu National Park
Alaru Creek	NT	106	Garig Gunak Barlu National Park
Breakneck River	SA	98	Ravine des Casoars Wilderness Protection Area
Yidney Creek	Qld	91	Great Sandy Conservation Park
Louisa River	Tas	88	Southwest National Park
Ilamaryi River	NT	87	Garig Gunak Barlu National Park
Manukalungku Creek	NT	62	Limmen National Park
Merrica River	NSW	61	Nadgee Nature Reserve
Nadgee River	NSW	60	Nadgee Nature Reserve
Obiworbbby Brook	NT	59	Kakadu National Park
Darby River	Vic	59	Wilsons Promontory National Park

9.2.2. Comprehensiveness and adequacy of the National Reserve System

The level of protection afforded by the National Reserve System varies greatly among River Environment Types (Figure 9.7). The protected area network contains more than 15% of the stream length of only a small proportion of River Environment Types. It represents many River Environment Types poorly or not at all, the more so the finer the partitioning of stream segments by the classification. Thus, the protected area network includes streams of all Level 1 types but less than 40% of Level 5 types. Many of the unrepresented Level 5 types,

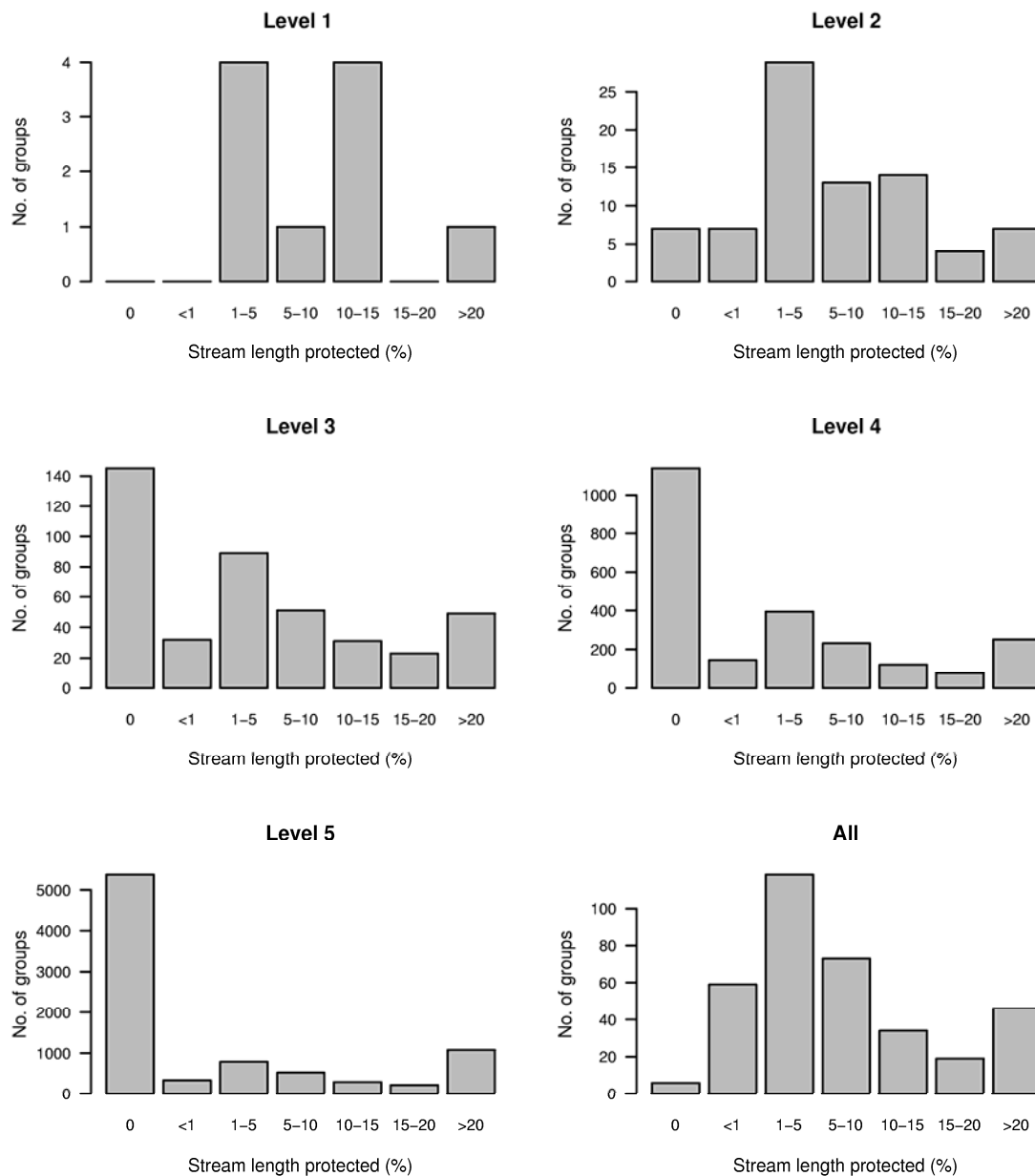


Figure 9.7 Proportion of stream length within an IUCN category I to IV protected area by River Environment Types computed for each of the levels in the hierarchical and the combined (ALL) classification

totalling 240,368km of stream, have a limited geographic extent. They exist for an average length of just 45km and only 731 types total more than 10km of stream. In contrast, only six of the ALL types with a total length of just 12,669km (average 2,112km) are not sampled by the NRS. Relatively few types occur entirely within protected areas - just 4, 36 and 199 of the types defined by the hierarchical classification Levels 3, 4 and 5 respectively. However, the River Environment Types that are included within the National Reserve System are usually sampled from more than one drainage basin though the relative extent of replication declines, as the number of types increases (Table 9.6). This is notwithstanding a parallel increase in the proportion of types that are unique to a drainage basin.

Representation of rare and threatened River Environment Types within the NRS also varies among the classifications. Rare River Environment Types are not well represented: 43% of rare ALL types and as many as three-quarters of rare Level 5 types are not found within a protected area (Table 9.7). The protected area system samples less than one-third of threatened or endangered Level 5 types but all endangered and threatened ALL types. For all but the hierarchical classification Level 1, “extinct” types were identified, that is, types where all of the stream length was moderately to highly disturbed ($RDI \geq 0.3$). Yet, some of these extinct types are found in protected areas. This apparent anomaly arises where there are flow regulating structures or intensive landuses upstream of the protected area.

Table 9.6 Replication within the National Reserve System. Numbers of River Environment Types protected in more than one drainage basin. Some types are unique to a drainage basin and therefore could not be replicated.

Classification		Number of types			
		Total	At least one example in a protected area	Protected and occurs in more than one drainage basin	Replicated
ALL		355	349	345	330
Hierarchical	Level 1	10	10	10	10
	Level 2	81	74	73	67
	Level 3	420	275	264	215
	Level 4	2361	1224	1152	826
	Level 5	8614	3242	2843	1745

Table 9.7 Numbers of rare and threatened River Environment Types and their representation within the National Reserve System. Assignment of type status is described in Section 9.1.4

Classification		Extinct		Endangered		Threatened		Rare	
		Total	NRS	Total	NRS	Total	NRS	Total	NRS
ALL		3	3	5	5	6	6	14	8
Hierarchical	Level 1	0	0	0	0	0	0	0	0
	Level 2	2	1	2	1	4	1	14	7
	Level 3	11	1	6	2	12	7	194	49
	Level 4	49	4	17	7	20	7	1541	416
	Level 5	395	41	51	17	116	30	6934	1585

Table 9.8 Numbers of River Environment Types where remaining unprotected and undisturbed stream length is insufficient to achieve the conservation target.

Classification		Number of types		
		Total	Requiring > 100% of undisturbed streams	With no undisturbed stream remaining
ALL		355	137	27
Hierarchical	Level 1	10	2	0
	Level 2	81	28	11
	Level 3	420	122	115
	Level 4	2361	516	867
	Level 5	8614	1167	4036

While the relative numbers of types in representation classes differs between the hierarchical and ALL classifications (Figure 9.7) broadly consistent geographic patterns emerge when representation levels are mapped as a proportion of the computed conservation targets (Figure 9.8). River Environment Types along the eastern divide and throughout much of Tasmania (excluding the Tasmanian midlands and northern coastal areas) are relatively over sampled by the National Reserve System. The NRS has also made substantial progress towards achieving conservation goals for River Environment Types across much of the Top End with some notable exceptions including those of the Mitchell, Norman and Flinders Rivers in the Gulf of Carpentaria region. In contrast, many more streams and their catchments will need to be added to the NRS if conservation targets for river ecosystems on the slopes and tablelands regions of southern Australia and across much of the interior of the continent are to be attained. In total, 380,056km or 406,709km of stream would have to be added to the length of already protected streams to reach conservation targets for the ALL and hierarchical Level 5 classifications, respectively.

Protecting remaining undisturbed examples of River Environment Types is far more cost effective than attempting to restore degraded systems (Natural Resource Management Ministerial Council (NRMMC), 2005). It is difficult, however, to find sufficient remaining undisturbed length of stream to recommend for inclusion in the NRS for many of the under sampled River Environment Types (Figure 9.9). Even if all of the undisturbed streams of some types were added to the protected area network, representation levels would still fall short of the targets for large numbers of River Environment Types (Table 9.8). For many types, no undisturbed stream remains. The River Environment Types associated with major rivers, including the Murray, Murrumbidgee, Darling, Fitzroy and Burdekin Rivers, feature prominently among those for which all unprotected streams are disturbed to some degree (Figure 9.9).

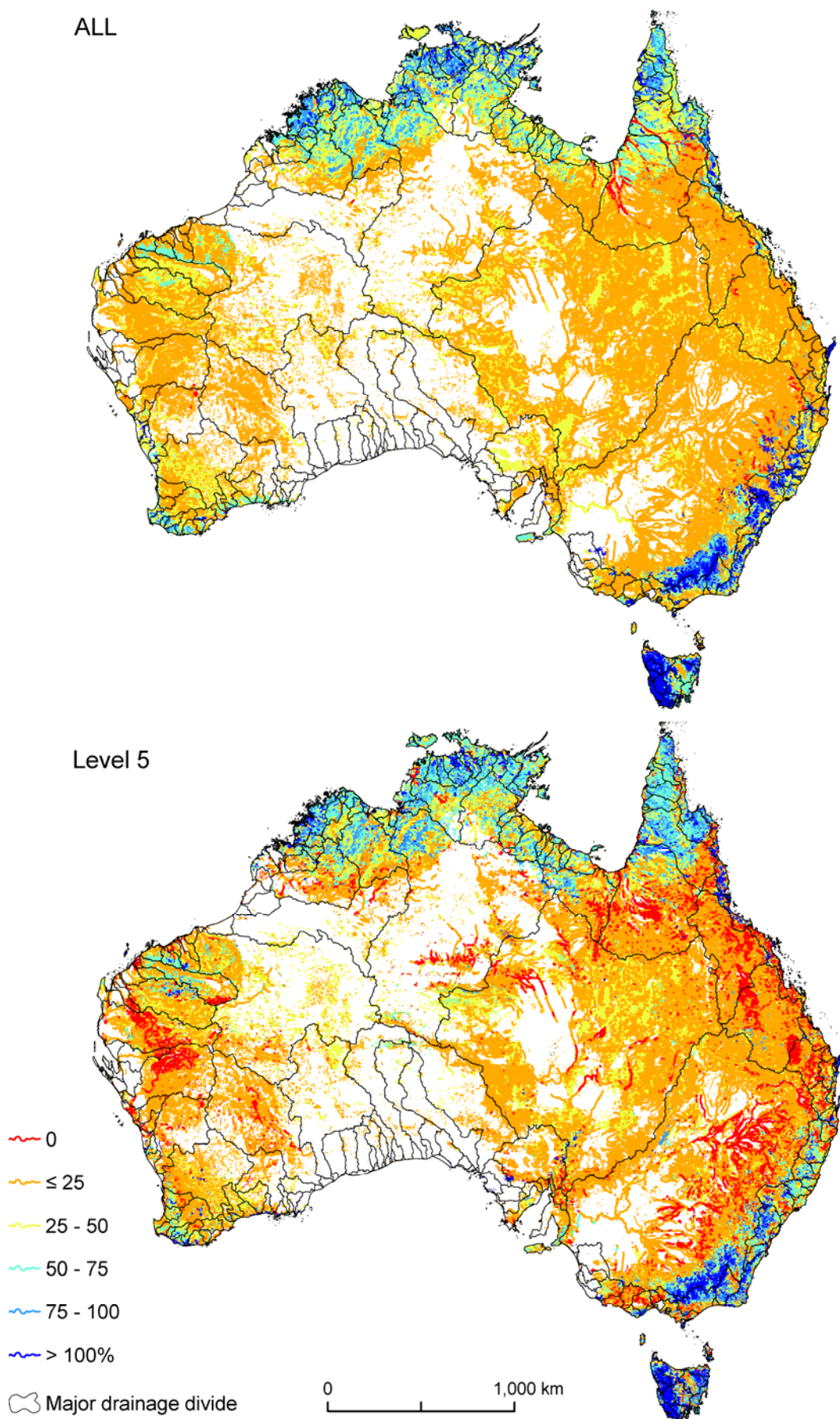


Figure 9.8 Progress towards conservation targets: length of stream within an IUCN category I to IV protected area as a percentage of target length for ALL and Level 5 River Environment Types

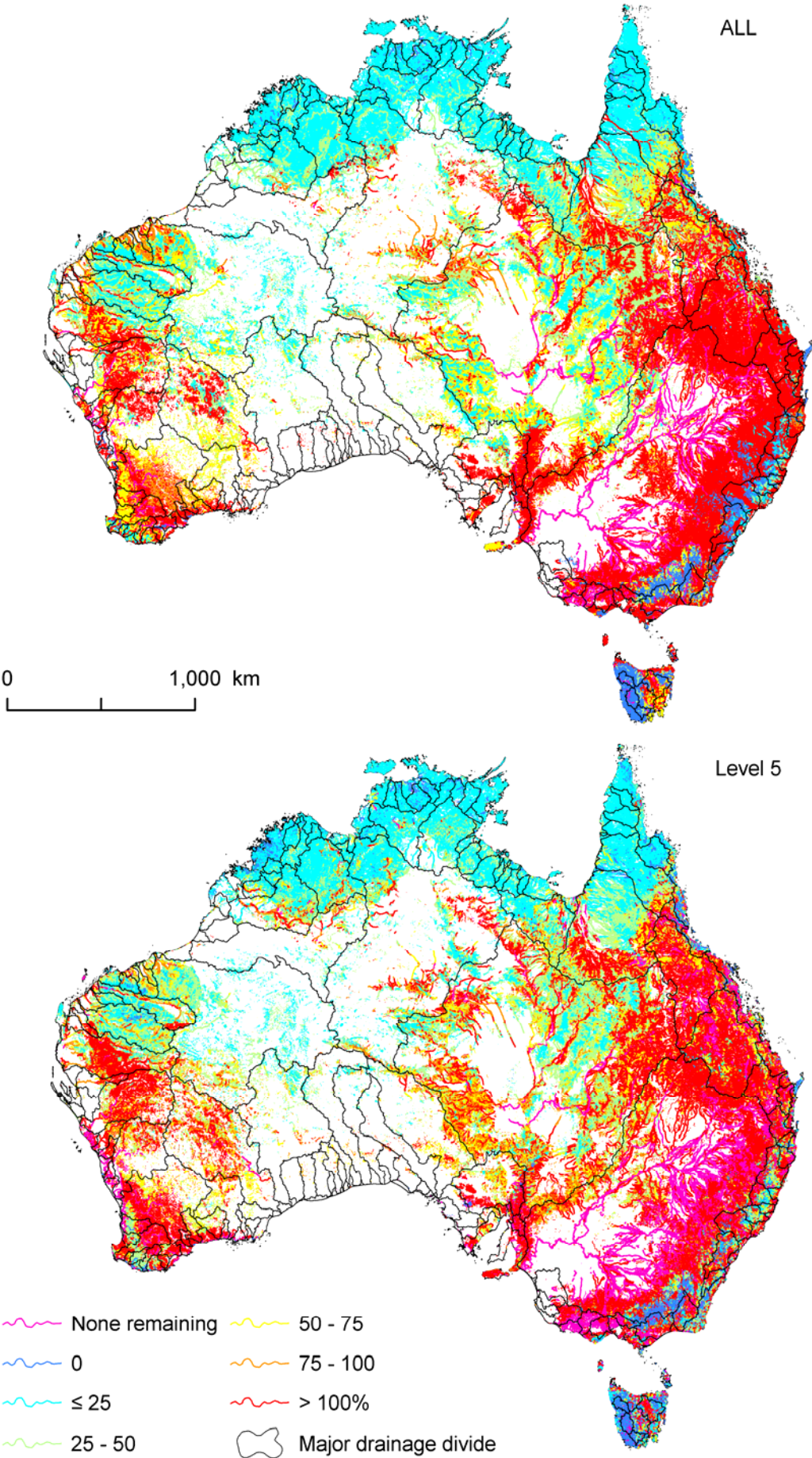


Figure 9.9 Proportion of remaining undisturbed (RDI < 0.05) and unprotected stream length required to achieve conservation targets for ALL and Level 5 River Environment Types.

9.3. Discussion

9.3.1. Comprehensiveness and adequacy

As at 2002 the IUCN management category I to IV protected areas that comprise the National Reserve System totalled 540,144 km² (Environment Australia, 2003), or 7% of the Australian landmass. These protected areas contain a similar proportion (6.7%) of the total stream length (at a map scale of about 1:250,000), but if the potential impacts of human activities within the catchment are accounted for, the protected proportion drops to less than 4%, emphasising the need to consider conservation measures holistically within a catchment framework. A figure of 4% is much less than that regarded as adequate for long-term conservation of natural ecosystems in Australia (e.g. JANIS (Joint ANZECC / MCFFA National Forest Policy Statement Implementation Sub-committee), 1997). Furthermore, these global summary statistics mask significant bias within the NRS. While a few river ecosystems are well protected many others have very limited or no protection. The results corroborate suggestions of a bias in the reserve system towards higher gradient streams in upper catchment areas (Tait *et al.*, 2003).

The stated objective of the NRS is a comprehensive, adequate and representative (CAR) protected area system across Australia, containing enough samples of all ecosystems identified at regional scales to ensure their ecological viability, resilience and integrity (Natural Resource Management Ministerial Council (NRMMC), 2005). Progress towards comprehensiveness is judged by the proportion of extant regional ecosystems represented within the NRS and adequacy by the degree to which protected areas replicate sampled ecosystems, are of sufficient size, managed within a bioregional framework and optimize opportunities for dispersal (Natural Resource Management Ministerial Council (NRMMC), 2005).

By these criteria, the NRS has some way to go to achieve its goal of a CAR protected area system for riverine ecosystems. At best, it samples 98% of River Environment Types (ALL), a seemingly good result. However, the ALL classification produced relatively heterogeneous groups that may not be suitably representative of particular river environments. Analysis of the more homogeneous hierarchical classification Level 5 types indicates a much bleaker picture for reserve system comprehensiveness with just 37% of types included within protected areas. Rare and threatened ecosystems are also poorly represented.

The adequacy of the NRS for the conservation of riverine ecosystems is very uncertain. While the extent of replication is reasonable, the size of the protected area network, as measured by the extent of stream length it contains, is generally well below target lengths. Moreover, the computed target lengths are likely to be conservative estimates of the length of stream that requires protection to ensure long term persistence of river ecosystems. The targets are adjusted

for variability in the natural rarity and vulnerability of River Environment Types but the minimum target value of 10% of stream length is essentially arbitrary and may not be enough to ensure a high likelihood of persistence for all ecosystem types. Credible targets would also consider the key hydrological, geomorphological and ecological processes that sustain river ecosystems. Adequacy would then be judged not only by the extent of protected streams but also by the degree to which existing protection measures also protect these critical processes. The NRS would appear to be even more inadequate had, for example, the stricter test of applying protected status only to those stream segments with fully protected upper catchments and not affected by any barriers to movement of aquatic biota, been imposed.

The CAR criteria require that areas included within a reserve system should also be representative, that they reasonably reflect the essential characteristics of the ecosystem and their inherent variability (Natural Resource Management Ministerial Council (NRMMC), 2005). The representativeness of streams already included within protected areas was not assessed but if desired could be readily determined by calculating the average distance in environmental space of each protected stream segment from its classification group's centroid (Belbin, 1993a).

9.3.2. Assessment reliability

Inevitably there will be uncertainty in the results presented here. They already lack currency. Data for protected areas declared after 2002 as part of the ongoing program to address gaps in the NRS were not available in time for this analysis. The recently released summary statistics for CAPAD 2004 indicate that more than 3 million hectares were added to the protected area system since CAPAD 2002 was compiled (CAPAD 2004, <http://eriss.erin.gov.au/parks/nrs/capad/2004/index.html>). Section 7.4.3.2 recognizes classification uncertainty arising from a variety of sources including the coarse resolution of spatial data for some landscape attributes. Consequently, without additional information it is not possible to be certain that River Environment Types that occur rarely are not simply artefacts resulting from the use of data of incompatible spatial scales. This is especially true for the hierarchical classifications.

Grid based overlay procedures were employed for this analysis because of their computational efficiency and compatibility with the format of the spatial framework and classifications but will introduce additional uncertainty. Gridding vector based spatial data produces boundary errors that affect the accuracy of computed areas and stream lengths. The grid representation might also erroneously indicate a stream is within or outside of a protected area where the stream forms the boundary of the area. However, while such errors might occasionally be locally significant they are unlikely to have a major bearing on conclusions concerning the comprehensiveness and adequacy of the NRS continent-wide.

The criteria that define rare and threatened ecosystems loosely follow those established for terrestrial systems but their applicability to rivers and streams is yet to be tested. The reliability of a stream segment's threatened or endangered status relies on the suitability of the RDI threshold value. This value is indicative only of the extent and intensity of the pressures on stream ecosystem processes, not a measure of its condition (Stein *et al.*, 2002). Moreover, a single value is unlikely to be appropriate for all River Environment Types and all combinations of anthropogenic disturbance. A measure of a River Environment Type's vulnerability is also derived from RDI values. The currency and reliability of these disturbance indices could be improved by a more extensive revision of the disturbance database and enhancements to the modelling procedures (Stein *et al.*, 2002).

9.3.3. Classification choice

The results for all classifications show there are significant gaps in the NRS but the details vary considerably. The choice of classification influences indicators of bias in the protected area network. For instance, relatively more River Environment Types were found to be unrepresented or inadequately represented for the classifications that define the larger number of types. The same is true for rare and / or threatened types.

This variation is not only due to the number of types recognized but is also attributable to the classification method. Although defining similar numbers of types, the hierarchical Level 3 classification types are relatively less often replicated and less comprehensively represented within the NRS than the ALL types. Many more Level 3 types are also rare. The properties of the groups produced by the alternative clustering approaches provide an explanation. The ALL classification, a single classification of stream segments based on their shared similarity across all landscape attributes, produces relatively heterogeneous River Environment Types but of more even group sizes than the hierarchical combination of multiple classifications. Because the ALL groups are generally larger they more often occur within in a protected area.

Classification choice will clearly influence perceptions of the comprehensiveness and adequacy of a protected area network and hence the need for conservation action. A simple prioritisation exercise further demonstrates its effect on conservation planning outcomes. In this example, stream segments and their sub-catchments (the planning units) are identified for priority protection if they represent River Environment Types for which almost all (> 90%) of the remaining undisturbed and unprotected stream length is required to achieve conservation targets. Prioritising the acquisition of planning units for inclusion where there are no alternatives that would equally contribute to achievement of conservation targets, reduces the likelihood that targets are compromised before a conservation plan is fully implemented (Margules and Pressey, 2000; Pressey and Taffs, 2001).

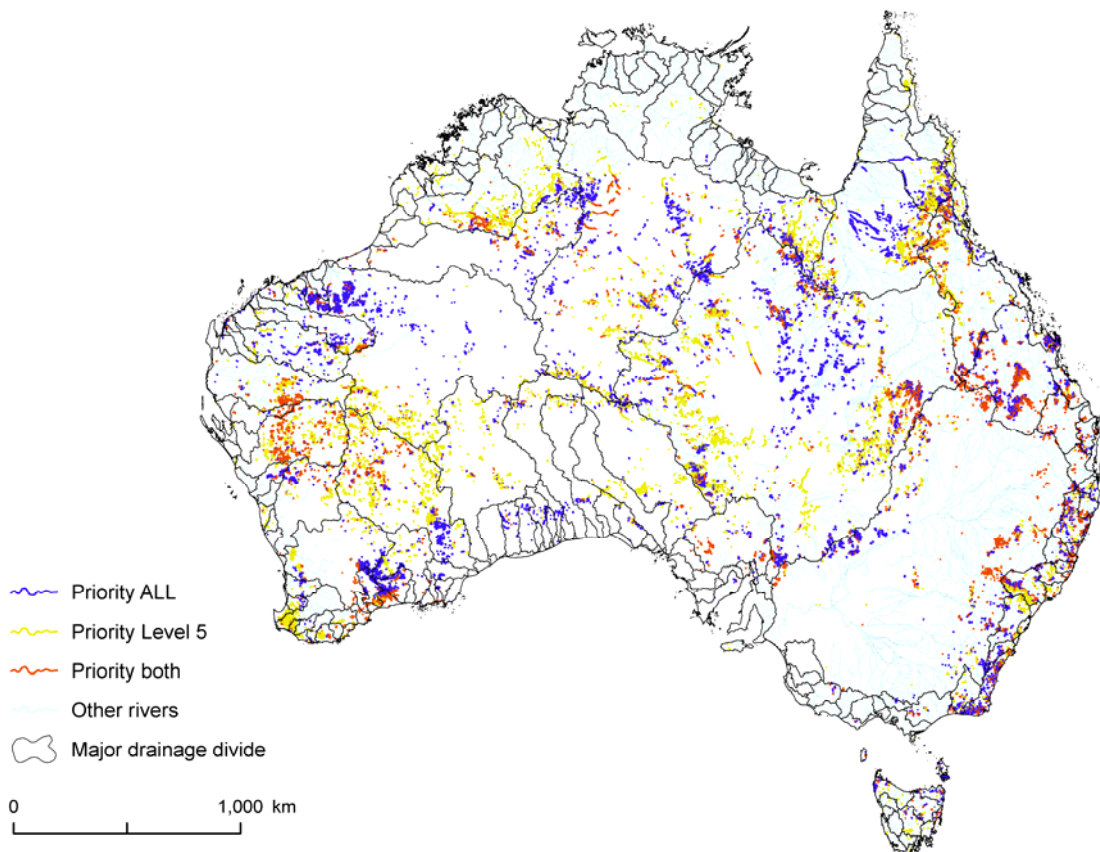


Figure 9.10 Priority streams for conservation action: more than 90% of the remaining undisturbed streams are required to achieve conservation targets for River Environment Types.

For the ALL River Environment Types some 69,787km of stream would be considered for priority protection because there are no undisturbed alternatives. The equivalent figure for the Level 5 types is slightly less at 66,311km but only 27% (29,173km) of the total length of priority streams identified was common to both classifications (Figure 9.10). In each case, about 40,000km of stream selected by one classification was not identified by the other. Notably, many of the priority streams identified are not within the NRS Program's highest priority IBRA regions (<http://www.deh.gov.au/parks/nrs/ibra/priority.html>). IBRA priority regions were determined according to their reservation status, comprehensiveness and the degree of threat to biodiversity (Natural Resource Management Ministerial Council (NRMMC), 2005).

9.4. Conclusions

The landscape framework components developed in this study did support a review of the existing protected area network, which is an essential stage in systematic conservation planning. Despite caveats with respect to both data and methodology, it is clear that the NRS does not yet deliver a comprehensive and adequate system of protected areas for the conservation of river ecosystems. Substantial gaps and inadequacies in the network of protected areas were identified. These may be difficult to address, more so if implementation of protection measures is delayed. Restoration or rehabilitation of degraded streams will be required if all conservation targets are to be achieved.

Chapter 10. Outcomes and conclusions

In this thesis I argue for a systematic, nationwide approach to conservation planning for Australia's rivers and streams. I address a fundamental impediment to the application of systematic conservation planning, namely, the lack of a suitable spatial framework and classification. Here, the outcomes of the study are examined. Both the landscape framework and its application are considered in Section 10.1. The uncertainty associated with these products was discussed in earlier chapters. In Section 10.2, the broader implications for conservation planning are explored and suggestions offered to improve the reliability of planning outcomes. Finally, the methodological contribution of this research and its wider applicability are considered (Section 10.3).

10.1. A continental landscape framework for Australian rivers and streams

10.1.1. Systematic conservation planning applications

The continental landscape framework developed in this study will facilitate a more consistent and comprehensive approach to conserving Australia's distinctive riverine ecosystems. The framework is built upon a multi-scale stream and nested catchment reference system that serves as a spatial framework. The structure of the spatial framework reflects the inherent hierarchical organization of a river system. Catchment units, the basis of this framework, are widely adopted as the basic planning units for river conservation programs in Australia and elsewhere (Section 2.1). Stream segments and their direct sub-catchments are characterised and classified according to the landscape controls that ultimately control river ecosystem patterns and processes. Indices of anthropogenic disturbance, also derived for individual stream segments, complete the continental landscape framework.

The nested catchment framework overcomes many of the shortcomings of existing catchment frameworks. It correctly depicts connections between catchment units via distributaries that may otherwise confound biogeographic analyses or prediction of the downstream impacts of human activities. In contrast to the AWRC delineation of Australia's River Basins (Australian Water Resources Council, 1976; and recent revisions AUSLIG, 1997; Geoscience Australia, 2003a), catchment boundaries are strictly topographically defined. Consequently, the catchment units truly represent the surface drainage characteristics of the Australian continent, irrespective of administrative or jurisdictional borders. Drainage basins that traverse jurisdictional borders may be assessed consistently regardless of administrative arrangements. The spatial framework

realistically delineates the extent of channelised flow and the natural variation in drainage density (at map scales of about 1:250,000), unlike the stream networks derived automatically with uniform contributing area thresholds (e.g. Hutchinson, Stein *et al.*, 2000).

The multiple scales of the spatial framework satisfy the requirements of a range of conservation planning tasks. Stream segments, the finest scale in the spatial framework, are the objects of the landscape classifications. They are also logical units to attribute biodiversity information including data on the distribution and abundance of species or communities, enabling greatest flexibility for evaluation and reporting. Data can then be aggregated at broader scales as appropriate. Stream segments and their direct sub-catchment areas may be used as the planning units for reserve design algorithms to systematically identify priority streams for conservation action. Larger sub-catchment units (e.g. a 5 or 6 level Pfafstetter sub-division) are likely to produce priority sets that represent biodiversity less efficiently, but with greater spatial contiguity and therefore easier to apply protective management measures (Section 2.2.1). In fact, sub-catchments at both scales could be used. Congruence between the different sets of priority streams would generate greater confidence in conservation planning recommendations (Warman *et al.*, 2004). Larger catchment units may also serve reporting needs better.

Meaningful visualisation of segment scale variation is best achieved within a GIS or using an online mapping system, but where information must be presented at small spatial scales to satisfy print requirements, data may be usefully generalised by aggregating the segment data within a higher level catchment unit.

The Pfafstetter coding that accompanies the nested catchments facilitates analysis of connectivities between sections of the drainage system. This property, not available in any other continental scale catchment framework in Australia, will encourage conservation strategies to consider the critical linkages that sustain riverine ecological processes (Ward *et al.*, 2001). This is essential if conservation measures are to be effective (Nevill, 2003; Kingsford *et al.*, 2005; Kingsford and Nevill, 2006).

Coupling the spatial framework with a classification makes possible many more conservation planning tasks. Chapter 9, for example, demonstrates the utility of the continental framework for systematic conservation planning with a review of the comprehensiveness and adequacy of the existing protected area system. The numerical dissimilarity value that quantifies the relationships between streams within and among River Environment Types allows reserve selection algorithms to choose streams that are most distinctive from those already within the protected area system (Faith and Walker, 1996) and those most representative of a type (Belbin, 1993a). The classification can assist in the assessment of conservation value criteria such as rarity, naturalness, diversity and representativeness because the attributes used to evaluate these criteria must be compared among rivers of the same type (Dunn, 2000). The landscape framework will, therefore, support alternative conservation planning approaches that use these

criteria to identify conservation importance (e.g. Boon *et al.*, 1998; Bennett *et al.*, 2002; Dunn, 2003). The classification also provides an effective basis for stratifying rivers for ecological survey and could assist in the identification of a reference state for assessment of ecological condition. Moreover, the consistent and comprehensive characterization of rivers that underpins the landscape classifications provides valuable input data for modelling the distribution of aquatic biota.

The continental landscape framework complements finer scale jurisdictional information such as that compiled for the Tasmania Conservation of Freshwater Ecosystems Values (CFEV) project (Corbett *et al.*, 2005; Davies, 2005; Dunn and Heffer, 2005; Long, 2005). It sets the context for the evaluation of national conservation significance and provides a consistent description of river systems that can enhance sharing and communication of information among planners. The landscape framework can encourage cooperation and coordination among State and Territory agencies, local governments and catchment management authorities, essential if a national conservation plan is to be effectively implemented. Agreement among State, Territory and Commonwealth nature conservation agencies over IBRA boundaries, for instance, was seen as a significant breakthrough for the Australian National Reserves Systems Cooperative Program (Pigram and Sundell, 1997).

The continental framework describes potential ecosystem states rather than necessarily current conditions and reflects the natural, topographically controlled drainage characteristics of the land surface. Thus it delineates the original drainage system of the upper Gordon and Huon Rivers in south-west Tasmania, though now flooded due to the construction of large dams for hydro-electricity production. While much of the runoff generated in the upper catchment of the Huon River is diverted via McPartlan Canal into Lake Gordon (<http://www.hydro.com.au/home/Energy/Tasmanian+Hydro+Electric+Schemes/Catchment+Areas/Gordon+Catchment/>), the drainage analysis of this study indicates that the catchments of the Gordon and Huon Rivers are disconnected drainage basins. Accordingly, conservation targets may be framed relative to pre-European distribution of ecosystems complying with the target setting principles set out in the directions paper for the National Reserve System (Natural Resource Management Ministerial Council (NRMMC), 2005)(Table 10.1).

With the addition of an indicator of anthropogenic disturbance, such as the River Disturbance Index, the conservation status of river ecosystems (represented by the River Environment Types) can be assessed, including the extent to which they have been lost or degraded. Rare, vulnerable and endangered ecosystems can then be identified for priority inclusion within the NRS as required by the NRS target setting principles. The NRS principles recommend that ecosystems be recognizable in the field. While the differences between the most similar River Environment Types may be quite subtle, they are, nonetheless, comprehensively mapped and because boundaries coincide with tributary confluences, should be readily located in the field.

Table 10.1 A continental landscape framework for Australia's rivers and streams: compliance with the target setting principles established for the National Reserve System (after Natural Resource Management Ministerial Council (NRMMC), 2005)

NRS target setting principle	Landscape framework for rivers and streams
Establishing targets based on some relativity with pre-European distribution is seen as a desirable NRS reservation objective.	Spatial framework and classifications are indicative of pre-European conditions
Priority attention in planning the NRS needs to be given to rare, vulnerable and endangered communities and species.	Identifies rare, vulnerable and endangered River Environment Types that potentially host rare, vulnerable and endangered communities and species (needs further investigation)
Ecosystem mapping at 1:100,000 or 1:250,000 scale is considered the appropriate scale for planning the NRS.	Spatial framework and most classifying attributes (though not substrate) derived from mapping at scales no coarser than 1:250,000 scale
Ecosystems need to be recognizable in the field, be mappable and able to have their pre-1750 or pre-clearing distribution modeled or mapped.	The ecosystems represented by the River Environment Types are comprehensively mapped and reflect natural ecosystem drivers free of post-1750 anthropogenic influences
Regional ecosystems are an important surrogate along with species information for planning the NRS	River Environment Types provide a reasonable surrogate, representing significant variation in riverine biotic communities and habitat characteristics
Biodiversity conservation objectives are best planned and delivered through the development of conservation strategies that integrate these approaches in a regional, catchment or landscape context	The hierarchical nested catchment framework facilitates integrated planning and delivery of conservation strategies in multiple spatial and landscape contexts
IBRA regions and subregions as outlined in IBRA V5, and subsequent updates, are the best planning framework for the NRS.	River Environment Types provide an alternative planning framework better representing the unique catchment constraints of river ecosystems

IBRA provides the basic planning framework for the NRS, although the Natural Resource Management Ministerial Council that is responsible for the NRS program acknowledge that it may not provide the best framework for freshwater biota (Natural Resource Management Ministerial Council (NRMMC), 2005). IBRA neither accounts for the catchment constraints that influence the distribution of obligate freshwater taxa (Unmack, 2001; Tait *et al.*, 2003) nor the critical linkages that comprise a drainage system. The landscape framework developed in this study overcomes these limitations. The spatial framework recognises linkages between elements of the drainage system and dispersal barriers in the form of catchment divides, while the classification describes patterns in the distribution of landscape variables that control stream ecosystem processes at multiple scales.

Another regionalisation, the Interim Marine and Coastal Regionalisation for Australia (IMCRA) (Interim Marine and Coastal Regionalisation for Australia Technical Group, 1998) supplies the planning framework for the National Representative System of Marine Protected Areas (NRSMPA) (<http://www.deh.gov.au/coasts/mpa/nrsmpa/index.html>). The separation of terrestrial, marine and freshwater systems however, is an artificial distinction that hinders conservation (Howitt, 2001; Abell, 2002), though without a common planning framework it is difficult to effectively integrate each of the biological realms into the conservation planning process (Cowling and Pressey, 2003). The catchment units that are the basis of the continental landscape framework are also logical units for terrestrial ecosystem conservation planning (Lewis *et al.*, 1991; Nix, 1997; Magnusson, 2001; Noss *et al.*, 2002). They delineate landscape elements connected by topographically driven flows of water and mineral nutrients that shape the distribution patterns of plants (e.g. Pringle and Tinley, 2003) and their associated fauna. Bathymetry is an important determinant of marine habitat (Roff and Taylor, 2000) so perhaps there is also a role for a corresponding marine “catchment” to support integrated conservation planning across all biological realms.

Reserves are the cornerstone of a conservation strategy. However, because of their large size and the multiple landuses they support, reservation at a whole of catchment scale is rarely a viable option for larger catchments or drainage basins. Moreover, a network of reserves may not be the most effective approach for conservation of very mobile fauna such as waterbirds that are dependent on highly dynamic floodplain wetland resources (Roshier *et al.*, 2002). The continental landscape framework developed in this study will also support planning for “off-reserve” protection mechanisms that could include land use planning, environmental flows and incentives (Kingsford *et al.*, 2005). For example, the topographically defined nested catchment framework facilitates recognition of multi-scale linkages between catchment processes and high conservation value assets in a basin, and thus more effective management of threatening processes. It will also supply a suitable spatial framework to help identify candidate rivers for listing under a proposed ‘Australian Heritage Rivers’ system, a ‘whole of basin’ protection

scheme that would conserve the integrity of largely unmodified drainage basins (Kingsford *et al.*, 2005).

10.1.2. Other applications

The continental landscape framework has applications beyond conservation planning. Demonstration of its broader applicability for river planning and management will encourage acceptance of the new framework. This is already underway. Data layers underpinning the landscape framework, though not yet formally documented and packaged, are assisting the activities of government agencies responsible for river management.

The Murray Darling Basin Commission used the data to help select sites for the Sustainable Rivers Audit (Murray-Darling Basin Commission, 2004). In Queensland, the Department of Natural Resources, Mines and Water is employing the consistent characterization of landscape attributes to help identify the key physical drivers shaping river ecosystem response in each of the state's freshwater bioregions. They have also used framework data to devise suitable criteria for regulation of mining activities within declared wild rivers catchments. The department is also evaluating the derived stream network and the indices of anthropogenic disturbance to help identify reference sites for their monitoring of river health bioassessment program.

Data layers have also been provided to assist in the modelling of the distribution of aquatic biota in New South Wales (Ivor Growns, NSW Department of Natural Resources, pers. comm. 21/3/06) and to the Commonwealth Department of Environment and Heritage to delineate the catchment areas of wetlands listed on the Directory of Important Wetlands. A new Land and Water Australia funded project is to begin shortly in collaboration with the Centre for Riverine Landscapes at Griffith University using the landscape attributes in association with gauging station data to produce a hydro-ecological classification for Australian rivers.

The landscape framework can also supply basic datasets, including drainage basin boundaries, for global initiatives such as the Global Water Systems Program (<http://www.gwsp.org>). It could provide the central framework and key data layers for an online mapping and information system for Australian rivers and streams to complement and desirably, integrate with, information systems for estuaries, OzEstuaries (<http://www.ozestuaries.org/>) and wetlands, the Australian Wetlands database (<http://www.deh.gov.au/water/wetlands/database/index.html>). Relevant data on current condition and existing uses and information on conservation and other values are distributed among many agencies and organizations. The common framework could act as a repository for this information, facilitating data sharing and integrated analysis of social and biophysical data at multiple scales.

The landscape framework would enable the cumulative effects of incremental developments, 'one of the most important and intractable problems' facing water managers (Nevill, 2003), to

be more effectively accounted for and provide catchment management authorities with a broader regional perspective for prioritising and targeting management activities. The spatial hierarchical framework and landscape characterization will also be useful to researchers seeking to understand the scale dependency of ecosystem patterns and processes (Poff, 1997; Mackey and Lindenmayer, 2001; Parsons *et al.*, 2004a), advance the understanding of landscape influences on riverine biota (Naiman *et al.*, 2002) or distinguish stream responses to human activities from the confounding influence of natural environmental gradients (Allan, 2004).

10.2. Spatial data uncertainty

The continental landscape framework will support diverse applications at regional to continental scales, though the outcomes may be sensitive to the spatial data uncertainty associated with its components. Spatial data uncertainty arises from many sources (Table 10.2) and is propagated through each step of the analysis. Earlier chapters considered, among others, errors in stream and catchment boundary placement and the effect of input data scale and classification method on the reliability of the defined River Environment Types. More comprehensive testing, preferably with purposefully sampled data, is needed to clarify the major sources of spatial data uncertainty in the landscape framework. Errors and inadequacies in the DEM, in particular, will be conveyed to all DEM derivatives (Van Niel *et al.*, 2004), including climate estimates, terrain parameters, catchment boundaries and stream networks. Naturally too, the placement of the boundaries between river types will be uncertain where variation is continuous so that there is a gradual transition between types rather than a sharp demarcation.

The sensitivity of conservation planning outcomes to spatial data uncertainty has not been widely assessed (Wilson *et al.*, 2004). This is in contrast to the many studies that have examined the implications of DEM errors for hydrological and geomorphological analysis (Thieken *et al.*, 1999; Walker and Willgoose, 1999; Yin and Wang, 1999; Wise, 2000; Gallant, 2001; Shortridge, 2001; Yang *et al.*, 2001; Aronica *et al.*, 2002). Nonetheless, we could expect that failure to acknowledge potential errors and uncertainty might lead to poor conservation planning decisions. It may be that scarce conservation resources are disproportionately allocated to conservation measures for rare but artefactual River Environment Types or conversely, that a truly rare stream ecosystem type is not detected simply because the spatial extent of the geological formation controlling its distribution was smaller than the minimum mapping unit.

Table 10.2. Sources of spatial data uncertainty (adapted from Stine and Hunsaker, 2001). The sources that are potentially most important for the landscape framework are indicated in italics.

Source	Implications	Examples
1. Measurement uncertainty		
Geometric/ projection errors	Location and boundary uncertainty	Transferring georeferenced points from a sphere to the planar map; map reprojection; digitizing errors
Human / Instrument error	Attribute uncertainty	Weather recording; remote sensing imagery; data entry errors
<i>Human bias</i>	Lack of consistency; Artefacts resulting from poor edge matching across mapsheets/ analysis regions; imposing boundaries on attributes that vary continuously	Mapping of categorical data e.g. soils, geology, vegetation type
Currency	Spurious relationships	Combining data from different periods; not accounting for temporal change
2. Grain		
<i>Scale / resolution</i>	Map unit heterogeneity; sub- grid variability; Inability to detect attribute of interest accurately; Number of “types” identified	Inappropriate resolution eg slopes underestimated by low resolution DEMs; low order stream channels not identified
<i>Combining data of different scales</i>	Overlay produces artefacts / non-existent combinations	Artefactual River Environment Types
3. Extent		
Spatial	Inability to recognize relationships	Local vs catchment/ regional scale processes
Temporal	Uncertainty over “natural” range of variability	Single “snap shots”
4. Analysis		
<i>Analytical procedure</i>	Interpretation uncertainty; misclassification	Choice of algorithm, dissimilarity measure; underlying assumptions
<i>Model error</i>	Predicted value uncertainty	Use of surrogates; model parameterisation; over-simplification or ignorance of key processes

10.2.1. Planning with uncertainty

Planners can reduce the risk of poor conservation decisions, even in the face of significant uncertainty. Uncertainty might be explicitly incorporated into conservation planning procedures (e.g. Halpern *et al.*, 2006; Moilanen and Wintle, 2006) or its impacts on planning outcomes communicated to decision makers, for instance with a sensitivity analyses (e.g. Shortridge, 2001). Higher resolution spatial information, field and local expert knowledge will be essential to verify and refine selected priorities.

More biodiversity is likely to be retained if priorities are identified for multiple components of biodiversity, including species and environmental diversity (Flather *et al.*, 1997; Cowling *et al.*, 2004; Higgins *et al.*, 2004; Pressey, 2004; Bonn and Gaston, 2005). A biogeographic

classification(s), reflective of the effects of historical processes on species distributions, would complement the River Environment Types. For instance, the Australian fish provinces of Unmack (2001), revised to accommodate the updated delineation of drainage basins (this study), could form the basis for macro-regions in a hierarchical classification (Tait *et al.*, 2003) with River Environment Types to characterize environmental variation within fish provinces. A zoogeographic regionalisation provides the context for abiotic classifications in the classification scheme presented by Higgins *et al.* (2005) for freshwater biodiversity conservation planning and used by the aquatic component of the Missouri gap analysis project (Sowa *et al.*, 2004). Similarly, biogeographic regions together with the River Environment Classification (Snelder and Biggs, 2002) have assisted the identification of freshwater systems of national importance in New Zealand (Chadderton *et al.*, 2004).

As Chapter 6 (Section 6.1) indicates, however, a biogeographic classification based on a single taxonomic group may not adequately capture the distribution patterns of other aquatic taxa and thus multiple regionalisations for different functional groupings of aquatic and semi-aquatic taxa might be required (e.g. reflecting the variable vagility of different freshwater taxa, Tait *et al.*, 2003). A single integrated classification would simplify communication and adoption, but could require that the boundaries between the regionalisations be arbitrarily reconciled (e.g. Doeg, 2001). There is, in reality, no 'right' or best classification and no practical need to define a single classification for conservation assessment and planning. Indeed, it is wise to employ as many biodiversity surrogates as possible (Margules and Pressey, 2000; Nix *et al.*, 2000) to maximise the representation of biodiversity among the streams prioritised for conservation action. River Environment Types and biogeographical regionalisations might be supplemented by biodiversity surrogates drawn from a variety of sources including predicted species and/or community distributions and continuous surfaces of compositional dissimilarity (for discussion and evaluation of different surrogates see, for example, MacNally *et al.*, 2002; Margules *et al.*, 2002; Faith *et al.*, 2004). Local expertise will be essential to supplement limited biodiversity data and may also be integrated into a systematic planning approach (e.g. Balram *et al.*, 2004).

10.2.2. Reducing uncertainty

The preceding measures reduce the likelihood that spatial data uncertainty will have adverse impacts on planning outcomes but there is also considerable scope to reduce uncertainty in the landscape framework components.

Building the landscape framework from more reliable and higher resolution data would improve its accuracy appreciably. The major upgrade of the DEM that underpins the framework, due for release in late 2006 (M.F.Hutchinson, CRES, Australian National University pers. comm. 16/3/06), will necessitate revision of the framework components. It corrects substantial numbers

of errors in source data and makes use of the latest enhancements to the ANUDEM algorithm (Hutchinson, 2006) (and see Section 10.2 below). This version of ANUDEM incorporates significant improvements to the drainage enforcement algorithm and avoids the drainage artefacts that previously arose from complex streamline data. It largely prevents encoding onto the grid of spurious interactions between streamlines and cliff lines and indicates where a stream falls over a cliff, so facilitating division of stream segments at major breaks in slope. A higher resolution global DEM, the Shuttle Radar Technology Mission (SRTM) DEM (Farr and Kobrick, 2000) is also now available for Australia at a grid spacing of 3 seconds of latitude and longitude (about 90m). With consistent sampling of the terrain and representation of sand ridge and other smaller scale local features, the SRTM DEM potentially provides more accurate definition of surface drainage pathways. However, drainage analysis applications are currently limited by errors and artefacts including voids (no data values) over water bodies (International Centre for Tropical Agriculture (CIAT), 2005), systematic errors due to the effects of the incidence angle of the radar beam (Gamache, 2004; Jarvis *et al.*, 2004) and especially in low relief areas, artefacts produced by random noise and variations in measured heights due to uneven tree cover (Gallant and Oliver, 2005). Major programs are underway to address these problems (e.g. International Centre for Tropical Agriculture (CIAT), 2005; Lehner *et al.*, 2006). In particular, a consortium between CRES, CSIRO Land and Water and Geosciences Australia is working to produce a new, higher resolution, national DEM for Australia. Investment in more consistent, higher resolution data for other critical landscape attributes, such as bedrock geology, will also help to reduce uncertainty in future revisions of the landscape framework.

Analytical methods that address uncertainty directly might also be considered. Fuzzy clustering methods, for instance, deal with the problem of indeterminate boundaries by reporting the partial group membership of objects for each class (Wilson and Burrough, 1999; Li, 2001). These methods (e.g. AutoClass Cheeseman and Stutz, 1996; Cook and Potts, 2002) can also account for known errors in the classifying attributes, effectively ensuring less certain attributes have less influence on the classification (Webb *et al.*, 2005). Because the necessary attribute data are already collated such methods could be readily tested.

The continental extent of the framework is defined around conservation needs, and not, as is often the case, pragmatic considerations such as the responsibilities of the commissioning agency. Still, the Australian riverine biota has biogeographical affinities beyond the continental limits, in particular with that of New Guinea e.g. waterbirds (Halse *et al.*, 1996; Kingsford and Norman, 2002) and fish (Unmack, 2001). To cover the full geographic and environmental range of Australian riverine biota, a revision of the framework might extend its coverage to New Guinea. Much of the essential spatial data has already been compiled (Nix *et al.*, 2000), albeit at a coarser scale (about 1km grid resolution) and with coverage for only the Papua New Guinea part of the island for some data layers.

10.3. Contribution to methodology

This study offers major contributions to spatial analysis methodology. It has developed new methods of drainage analysis that are suitable for continental scale application and that account for diverse drainage structures. A significant feature of the new procedures, the coding of a second flow direction at flow bifurcations in order to represent distributary drainage structures, is now incorporated into ANUDEM (version 5.2 Hutchinson, 2006). However, coding bifurcations into the flow direction grid complicates regular GIS tasks such as delineating catchments, calculating catchment area or deriving stream order. The procedures that were developed to overcome these difficulties could assist other studies employing the new drainage analysis methods.

Enhancements to the internationally adopted Pfafstetter system are also proposed. In particular, the use of runoff, rather than catchment area to distinguish main stem and tributaries could assist the broader acceptance of the scheme as the results conform more closely to common conventions. This study contributes new ideas to emerging methods of landscape classification of rivers (Snelder and Biggs, 2002; Jerie *et al.*, 2003; Higgins *et al.*, 2005) and their conservation planning application (Chadderton *et al.*, 2004; Sowa *et al.*, 2004; Davies, 2005). It introduces new ways of characterizing the influence of landscape factors on river ecosystems (e.g. the catchment water balance) and evaluates alternative classification algorithms and organizational structures. It also identifies major research questions for classification methodology (e.g. hierarchical structure and attribution of critical landscape factors). With the review of the NRS, this study advances understanding of the influence of classification choice on conservation planning outcomes.

The methods applied in this study are generic. This is important, as rivers are one of the most imperilled of global ecosystems (Abell, 2002). The classification approach adopted here relies on information that is widely accessible or is supplied by global databases. The WorldClim database (Hijmans *et al.*, 2005; Hijmans *et al.*, 2006), for example, provides global coverage of long-term mean monthly climatic attributes at a grid resolution of 1km although calculation of some flow regime descriptors such as variability requires actual monthly values as used in this study. Likewise, the drainage analysis procedures could be implemented almost anywhere as they depend only on data normally available from topographic mapping or remote sensing sources (Farr and Kobrick, 2000). Accordingly, these methods could be applied to develop a landscape framework for the rivers and streams of other continents. Such a framework would help to fill critical data gaps that, worldwide, hinder integrated management of river ecosystems and a reduction in the rate of biodiversity loss (Revenga *et al.*, 2005).

10.4. Conclusions

This thesis has addressed its aims. A continent-wide spatial framework applicable to the distinctive character of Australian river systems has been developed at 1:250,000 scale. Landscape classifications of these rivers were produced and evaluated. Almost certainly, this is the first time that the streams of an entire continent have been classified at such fine scales. The utility of the derived landscape framework was demonstrated with a review of the National Reserve System. The landscape framework fills major gaps in the spatial data upon which systematic conservation planning approaches are crucially dependent.

Framework errors and limitations were acknowledged and these may produce uncertain planning outcomes. While it is vital that this uncertainty, and its potential impacts, be communicated to planners and decision makers, its existence is not reason for inaction. The conservation need is urgent. The threats to river ecosystems are profound, yet existing conservation initiatives are neither comprehensive nor adequate. The development of the landscape framework presents an opportunity to begin the task of preparing a national conservation plan for Australia's rivers and streams. A systematic conservation planning approach will ensure limited conservation funds are strategically allocated and conservation decisions are accountable.

Many areas for improvement of the framework have been identified. Others will arise, guided by the experience of users and evaluation of its performance. An ongoing program of review and revision of conservation plans will, therefore, be necessary as further research and development of the landscape framework is undertaken.

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Appendix 1. Separating valley floors from hillslopes: the Flatness Index

The ability to separate valley floors from hillslopes was critical for the identification of spurious channel heads (Section 4.2.2.1) and the derivation of several of the terrain attributes used to classify streams (Section 7.2.2.1). This was achieved by assigning Flatness Index classes to grid cells according to the values of two complementary topographic indices, the multi-resolution Valley Bottom Flatness Index (mrVBF) and the multi-resolution Ridge Top Flatness Index (mrRTF) (Gallant and Dowling, 2003). These indices differentiate erosional and depositional areas in the landscape at a range of scales depending on their areal extent, relative position and slope.

By applying suitable thresholds to the index values (John Gallant, CSIRO Land and Water, pers. comm. 16/4/2004, Gallant and Dowling, 2003), valley bottoms were separated from hillslopes and ridge top flats (Table A. 1). The index values usually classify most grid cells as clearly ridge top flat, valley bottom or hillslope. The class of grid cells with ambiguous values was inferred from the index with the larger value where the difference was at least two units. For smaller differences the dominant character of a location cannot be reliably determined and these grid cells have an indeterminate classification (John Gallant, CSIRO Land and Water pers. comm. 16/4/2004).

Flatness Index classes were derived from the values of the mrVBF and mrRTF indices computed from the national 9 second DEM version 2.1 (Section 4.2.1.1) using the algorithm of Gallant and Dowling (2003) (Figure A.1). Valley bottoms and hillslopes are equally represented continentally (Table A. 1) though in areas of higher relief, valley bottoms are found only as small, isolated pockets. In contrast, over large parts of the inland and the Gulf of Carpentaria they form vast floodplains and hillslopes are rare. Here too the character of many cells was not clearly defined.

Table A. 1 Flatness Index class definition and abundance. Area was computed as a proportion of the total continental area.

Class	Definition	Proportional area (%)
Hillslope (Erosional)	$\text{mrVBF} < 2.5$ and $\text{mrRTF} < 2.5$	34
Valley bottom	$(\text{mrVBF} - \text{mrRTF}) > 2$	34
Ridge top flat	$(\text{mrRTF} - \text{mrVBF}) > 2$	10
Indeterminate	$\text{abs}(\text{mrRTF} - \text{mrVBF}) \leq 2$	22

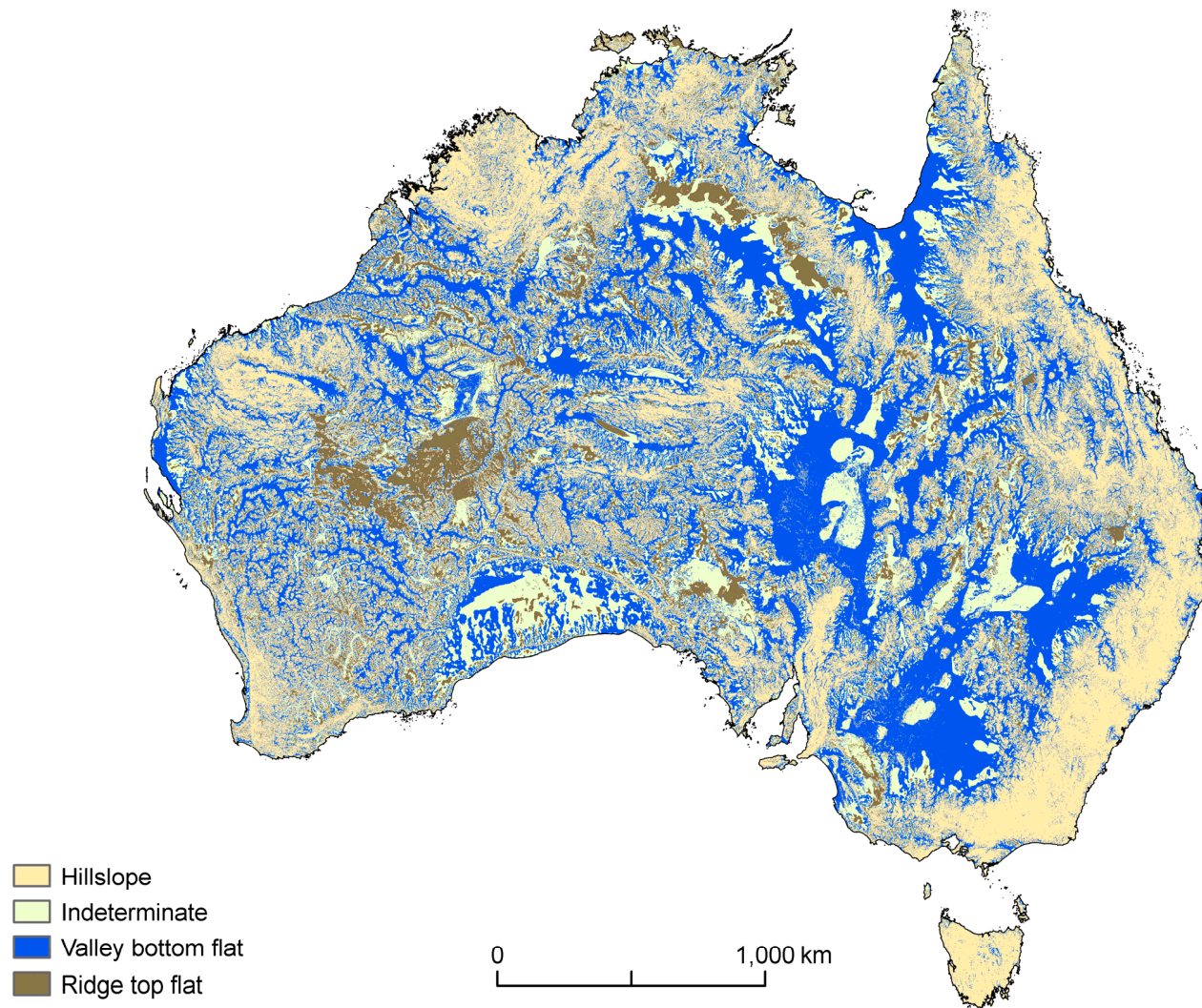


Figure A.1 Separating valley floors from hillslopes: Flatness Index Classes based on the values of the multi-resolution Valley Bottom Flatness Index (mrVBF) and multi-resolution Ridge Top Flatness Index (mrRTF) (Table A. 1)

Appendix 2. Comparing the catchment water balance with observed flow

A simple monthly catchment water balance is used to represent the most important controls on flow magnitude and variability, key facets of the flow regime (Section 7.2.2.1). In combination with other critical landscape attributes, summary statistics derived from the monthly water balance are expected to identify groupings of stream segments that display similar hydrological characteristics. To determine which statistics are reasonably estimated from the water balance computed with the GROWEST model, the water balance estimates were compared to those derived from observed monthly runoff for a national set of gauging stations assembled for the NLWRA extension of unimpaired streamflow project (Peel *et al.*, 2000) (Figure 4.12). This dataset includes monthly observations for 331 gauging stations on unregulated streams with a catchment area of between 50 and 2000km² and at least 10 years of record. Although it is biased towards smaller catchments in south-eastern Australia, it nonetheless provides a more reliable set for comparison than the more extensive series of stations compiled for the Australian Water Resources Assessment (National Land and Water Resources Audit, 2001b) that contains many missing records and inconsistent coding.

Statistical measures of the flow regime were calculated from the catchment water balance estimates derived for the stream segment of each gauging station for the years 1968 to 1998 and plotted against the same statistics computed for the observed monthly runoff values converted to a flow volume by multiplying by the gauging station's catchment area. Data from forty-five gauging stations that were not modelled adequately by the NLWRA extension of unimpaired streamflow project (Peel *et al.*, 2000) were removed from this analysis. Catchment water balance statistics are always based on the full 31 years of monthly data but the observed statistics may use fewer months of data as the NLWRA project only required gauging stations to have at least 120 months of recorded data (Peel *et al.*, 2000).

Even so, catchment water balance statistics are generally well correlated with observed monthly values of statistics describing the magnitude of annual, seasonal and high flows and intra- and inter-year variability (Figure A.2). Coefficient of variation values for the monthly and annual water balance overestimate the degree of variability but correctly identify the most variable streams in most cases. The catchment water balance produces winter mean values that are higher than those observed, in particular for several gauging stations in Victoria likely to be affected by delayed runoff from snow melt. In contrast, summer mean water balance estimates are more often lower than measured streamflows. For many gauging stations the catchment water balance erroneously predicts no summer flows.

This is not surprising given the use of monthly rainfall totals rather than actual rainfall events to model runoff and the failure of the model to differentiate quick flow and slow flow runoff

contributions to streamflow. As a consequence, the water balance is unable to produce meaningful measures of low flows. The minimum and 10th percentile of the monthly water balance estimates, for example, are zero for all or almost all gauging stations respectively. Similarly, while median values, an indicator of low flow conditions in many Australian streams (Young, Rustonji *et al.*, 2001), are reasonably predicted by the catchment water balance for streams with higher median flows, they are erroneously estimated as zero for more than half of the gauging stations. In fact, the median value of the measured monthly flows is zero for only two gauging stations.

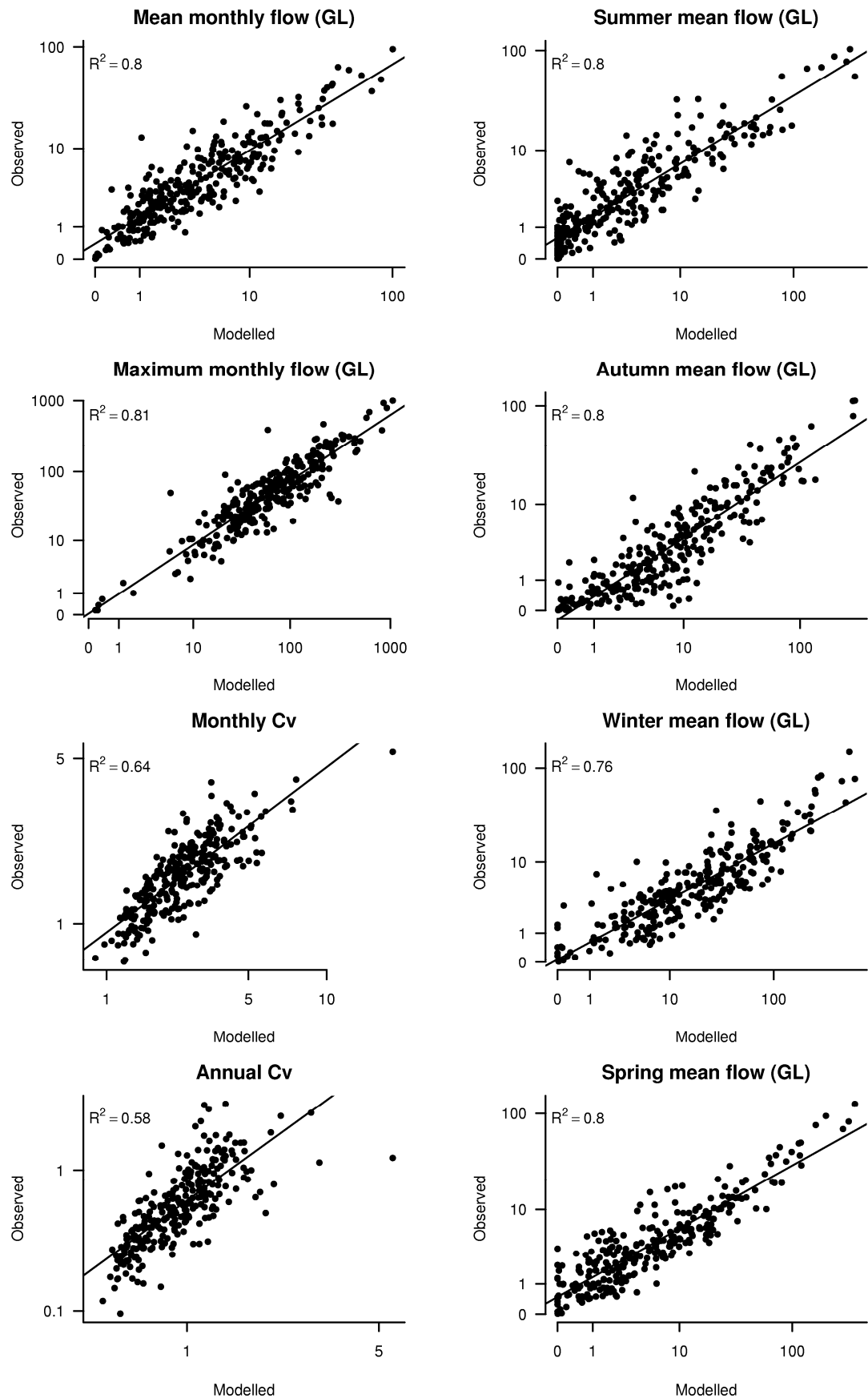


Figure A.2 Log-log plots of catchment water balance with observed flow summary statistics for gauging stations assembled for the NLWRA Extension of Monthly Mean Flow project (Peel *et al.*, 2000). Plots show linear regressions and adjusted R^2 values.

Appendix 3. Membership of ALL meta-groups

Table shows the ALL classification groups membership of meta groups derived by hierarchical agglomerative clustering of the ALL group centroids. The group member listed first for each meta-group is used to label the meta-groups in the dendrogram (Figure 7.18).

Meta group number	No. members	ALL members (group number)
1	23	1 2 4 6 7 8 9 10 12 13 14 15 18 19 20 21 23 31 45 48 90 98 105
2	6	64 81 93 99 106 341
3	4	25 26 39 202
4	34	3 5 24 30 32 34 35 41 44 51 59 62 82 86 87 88 94 96 110 165 336 337 342 132 175 100 156 103 111 124 140 181 149 210
5	23	17 22 27 37 38 40 42 47 54 56 57 66 67 69 73 79 109 131 144 221 239 339 343
6	7	46 68 72 75 77 78 340
7	2	70 338
8	30	43 61 65 80 89 107 114 116 117 119 121 123 129 135 147 154 171 178 182 183 189 211 214 134 138 158 238 204 232 244
9	22	50 58 63 71 104 108 122 150 153 157 161 180 190 203 222 230 257 275 283 303 349 350
10	4	245 246 250 282
11	14	74 115 118 133 141 142 148 155 167 169 185 200 227 347
12	34	76 83 84 120 127 130 137 145 146 152 168 173 174 176 177 179 187 188 231 241 247 253 352 91 235 92 95 215 242 102 151 139 159 345
13	11	11 16 29 33 36 53 60 97 113 125 128
14	14	28 49 52 55 85 101 112 126 136 186 205 213 219 223
15	13	143 166 170 194 199 207 226 251 255 289 292 298 308
16	7	249 268 279 295 307 318 354
17	2	260 266
18	4	212 263 270 276
19	5	160 201 217 273 346
20	31	163 164 172 184 191 193 195 196 197 198 208 216 218 237 243 248 259 281 286 290 291 301 323 209 262 267 317 261 309 280 325
21	12	233 284 285 294 297 299 300 302 304 306 334 351
22	20	162 220 224 225 228 229 234 236 240 252 254 256 258 264 265 269 271 296 305 348
23	8	192 206 278 288 314 319 326 344
24	15	277 287 293 311 313 315 316 320 321 322 331 332 333 335 355
25	10	272 274 310 312 324 327 328 329 330 353

Appendix 4. Field data sheets

The AusRivAS Physical Assessment field data sheets (after Parsons *et al.*, 2002) used for the survey of Northern Territory rivers. The USEPA Habitat Assessment on pages 8 to 11 of the data sheets was not attempted.

Date _____ Site No. _____ Time _____ Recorder's Name _____

River Name _____ Location _____

Weather _____ Rain in last week? Y [] N [] Photograph numbers and details _____

Latitude:	deg	min	sec	Longitude:	deg	min	sec
GPS Name and Datum _____							

PLANFORM SKETCH OF SITE

Including bedform types, location of cross-sections, access points, landmarks and natural or artificial channel or floodplain features.
Left bank is facing downstream.

LENGTH OF SAMPLING SITE

Bankfull width _____ (m)

x 10

Length of sampling site _____ (m)

Notes

BEFORE LEAVING THE
SITE, CHECK DATA
SHEETS TO ENSURE
THAT ALL VARIABLES
HAVE BEEN RECORDED

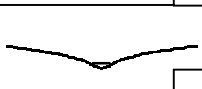
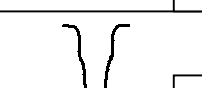
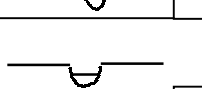
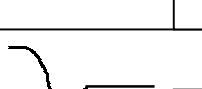
☐
Y

BASIC WATER CHEMISTRY

		Units
Temperature	_____	°C
Conductivity	_____	_____
Dissolved Oxygen	_____	mg l ⁻¹
Dissolved Oxygen Sat.	_____	%
pH	_____	
Turbidity	_____	
Total phosphorus ☐	_____	
Total nitrogen ☐	_____	
Water sample taken?		
ALKALINITY		
Amount of water	_____	ml
Amount of H ₂ SO ₄	_____	ml
Alkalinity	_____	mg l ⁻¹

Valley shape

Choose one category only

	Steep valley ☐
	Shallow valley ☐
	Broad valley ☐
	Gorge ☐
	Symmetrical floodplain ☐
	Asymmetrical floodplain ☐

Local impacts on streams

Choose one or more categories and describe the detail of each

☐ Sand or gravel mining	☐ Sewage effluent
☐ Other mining	☐ Channel straightening
☐ Road	☐ River improvement works
☐ Bridge / culvert / wharf	☐ Water extraction
☐ Ford / ramp	☐ Dredging
☐ Discharge pipe	☐ Grazing
☐ Forestry activities	☐ Litter
☐ Sugar mill	☐ Recreation
☐ Irrigation run-off or pipe outlet	☐ Other

 Description _____

Local landuse

Choose one category for each bank

Left	Right
☐	☐ Native forest
☐	☐ Native grassland (not grazed)
☐	☐ Grazing (native or non-native pasture)
☐	☐ Exotic grassland (lawns etc., no grazing)
☐	☐ Forestry Native [] [] Pine [] []
☐	☐ Cropped Rainfed [] [] Irrigated [] []
☐	☐ Urban residential
☐	☐ Commercial
☐	☐ Industrial or intensive agricultural
☐	☐ Recreation
☐	☐ Other _____

Floodplain width _____ Average _____ (m)

Floodplain features

Choose one or more features when present

☐ Sampling site has no distinct floodplain	☐ Scroll systems Short, crescentic strips or patches formed along the inner bank of a stream meander
☐ Oxbows / billabongs Body of water occupying a former river meander, isolated by a shift in the stream channel	☐ Splays Small alluvial fan formed where an overloaded stream breaks through a levee and deposits material on the floodplain
☐ Remnant channels Formed during a previous hydrological regime. May be infilled with sediment	☐ Floodplain scours Scour holes formed by the concentrated clearing and digging action of flowing water
☐ Flood channels A channel that distributes water onto the floodplain and off the floodplain during floods	☐ No floodplain features present Floodplain present at the sampling site but does not contain any of the above features

Riparian zone composition

Assess for whole sampling site

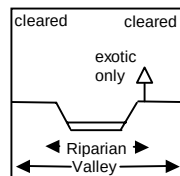
	% Cover		Vegetation Description
Trees (>10m in height)	_____	} May total more than 100%	_____
Trees (<10m in height)	_____		_____
Shrubs	_____		_____
Grasses / ferns / sedges	_____		_____

Shading of channel
☐ < 5% ☐ 6 – 25% ☐ 26 – 50% ☐ 51 – 75% ☐ > 76%
Extent of trailing bank vegetation
☐ nil ☐ moderate
☐ slight ☐ extensive
Native and exotic riparian vegetation

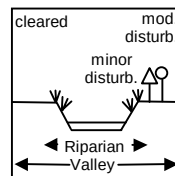
% Native _____ } Total 100%
 % Exotic _____

Overall vegetation disturbance rating

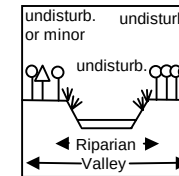
Choose one category only. Sites with valley vegetation cleared on BOTH sides, but with riparian vegetation in good condition should be scored in the high disturbance category. Words within the drawings summarise the detailed text about the state of the riparian and valley vegetation for each category.

Extreme disturbance ☐

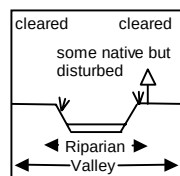
Riparian vegetation – absent or severely reduced. Vegetation is extremely disturbed (ie. dominated by exotic species with native species rare or completely absent)
Valley vegetation – agriculture and/or cleared land BOTH sides. Plants present are virtually all exotic species (willows, pines etc.)

High disturbance ☐

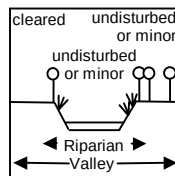
Riparian vegetation – moderately disturbed by stock or through the intrusion of exotic species, although some native species remain
Valley vegetation – agriculture and/or cleared land ONE side, native vegetation on the other side clearly disturbed or with a high percentage of introduced species present

Low disturbance ☐

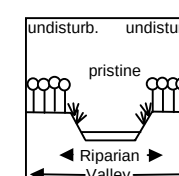
Riparian vegetation – native vegetation present on BOTH sides of the river and in relatively good condition with few exotic species present. Any disturbance present is relatively minor.
Valley vegetation – native vegetation present on BOTH sides of the river, with a virtually intact canopy and few exotic species

Very high disturbance ☐

Riparian vegetation – some native vegetation present, but it is severely modified BOTH sides by grazing or the intrusion of exotic species. Native species severely reduced in number and cover.
Valley vegetation – agriculture and/or cleared land BOTH sides. Plants present are virtually all exotic species (willows, pines etc.)

Moderate disturbance ☐

Riparian vegetation – native vegetation on BOTH sides with canopy intact or with native species widespread and common in the riparian zone. The intrusion of exotic species is minor and of moderate
Valley vegetation – agriculture and/or cleared land on ONE side, native vegetation on the other in reasonably undisturbed state

Very low disturbance ☐

Riparian vegetation – native vegetation present on BOTH sides of the river and in an undisturbed state. Exotic species are absent or rare. Representative of natural vegetation in excellent condition
Valley vegetation – native vegetation present on BOTH sides of the river with an intact canopy. Exotic species are absent or rare. Representative of natural vegetation in excellent condition

Longitudinal extent of riparian vegetation

Choose one category for each bank. Do not include ground layer except where site is in native grassland.

		Left bank	Right bank
None		☐	☐
Isolated / scattered		☐	☐
Regularly spaced		☐	☐
Occasional clumps		☐	☐
Semi-continuous		☐	☐
Continuous		☐	☐

Regeneration of native woody vegetation

Is the sampling site in undisturbed forest?

Y [] N []

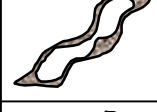
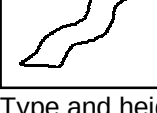
If no, record
regeneration
category

☐
☐
☐

Abundant (>5% cover) and healthy
 Present
 Very limited (<1% cover)

Physical barriers to local fish passage

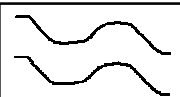
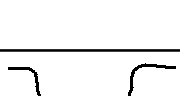
Choose one category for each flow condition

		Base flow	Low flow	High flow
	No passage	☐	☐	☐
	Very restricted passage	☐	☐	☐
	Moderately restricted passage	☐	☐	☐
	Partly restricted passage	☐	☐	☐
	Good passage	☐	☐	☐
	Unrestricted passage	☐	☐	☐

Type and height of barrier(s) _____

Type of bars

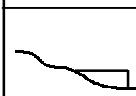
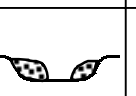
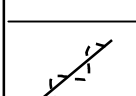
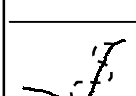
Choose one or more categories

	Bars absent	☐
	Side/point bars VEGETATED	☐
	Side/point bars UNVEGETATED	☐
	Mid-channel bars VEGETATED	☐
	Mid-channel bars UNVEGETATED	☐
	Bars around obstructions	☐
	Braided channel	☐
	Infilled channel	☐
	High flow deposits	☐

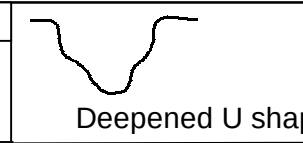
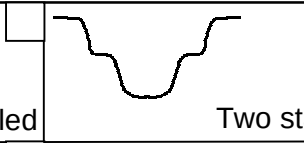
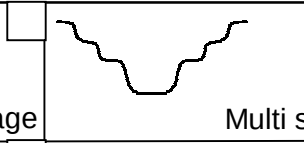
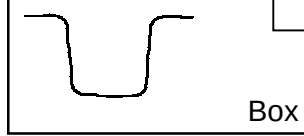
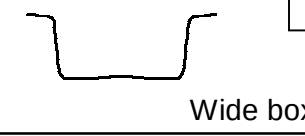
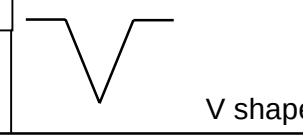
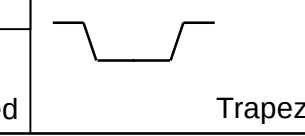
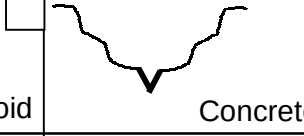
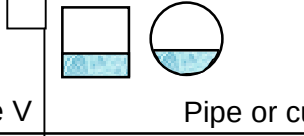
Extent of bars

% of streambed forming a bar of any type _____ %

Dominant sediment particle size on bars
 Boulder/cobble [☐] Pebble [☐] Gravel [☐]
 Sand [☐] Silt/clay [☐] or _____ mm
Channel modifications Choose one or more categories

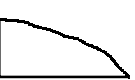
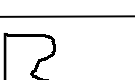
	No modifications	☐		Reinforced	☐
	Desnagged	☐		Revegetated	☐
	Dams and diversions	☐		Infilled	☐
	Resectioned	☐		Berms or embankments	☐
	Straightened	☐	Signs of work still	Recently channelised	☐
	Realigned	☐	Works old and revegetated	Channelised in the past	☐

Channel shape Choose one category only

	U shaped	☐		Flat U shaped	☐		Deepened U shape	☐		Widened or infilled	☐		Two stage	☐		Multi stage	☐
	Box	☐		Wide box	☐		V shaped	☐		Trapezoid	☐		Concrete V	☐		Pipe or culvert	☐

Bank shape

Choose one category for each bank

		Left bank	Right bank
	Concave	☐	☐
	Convex	☐	☐
	Stepped	☐	☐
	Wide lower bench	☐	☐
	Undercut	☐	☐

Factors affecting bank stability

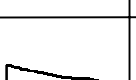
Choose one or more categories

- | | |
|--|---|
| ☐ None | ☐ Cleared vegetation |
| ☐ Mining | ☐ Irrigation draw-down |
| ☐ Runoff | ☐ Reservoir releases |
| ☐ Stock access | ☐ Seepage |
| ☐ Human access | ☐ Flow and waves |
| ☐ Ford, culvert or bridge | ☐ Drainpipes |
| ☐ Feral animals | |
| ☐ Other | |

 Description _____

Bank slope

Choose one category for each bank

		Left bank	Right bank
	Vertical 80 - 90°	☐	☐
	Steep 60 - 80°	☐	☐
	Moderate 30 - 60°	☐	☐
	Low 10 - 30°	☐	☐
	Flat <10°	☐	☐

Bedrock outcrops

Assess % of each bank covered by bedrock outcrops

 % bedrock outcrops Left bank _____
 Right Bank _____
Artificial bank protection measures

Choose one or more categories

- | | |
|--|---|
| ☐ None | ☐ Fenced stock watering points |
| ☐ Fence structures | ☐ Vegetation plantings |
| ☐ Levee banks | ☐ Logs strapped to bank |
| ☐ Rock or wall layer | ☐ Concrete channel lining |
| ☐ Rip rap | |
| ☐ Fenced human access | |
| ☐ Other | |

Sediment oils
☐ absent ☐ light ☐ moderate ☐ profuse
Water oils
☐ none ☐ flecks ☐ globs ☐ sheen ☐ slick
Sediment odours
☐ normal/none ☐ sewage ☐ petroleum ☐ chemical
☐ anaerobic ☐ other _____
Water odours
☐ normal/none ☐ sewage ☐ petroleum ☐ chemical
☐ other _____
Turbidity (visual assessment)
☐ Clear ☐ Slight ☐ Turbid ☐ Opaque

 ↓
 Is water clarity reduced by:

☐ Suspended material (e.g mud, clay, organics) ☐ Dissolved material (e.g plant leachates)
Water level at the time of sampling
☐ Dry ☐ No flow ☐ Low ☐ Baseflow or near baseflow
☐ High ☐ Flood (don't sample)
Artificial features at the sampling site

Choose one or more categories

☐ Major ☐ Minor ☐ Ford ☐ Bridge ☐ Culvert ☐ Other weir

 Description _____

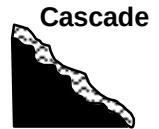
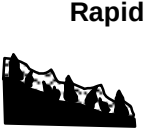
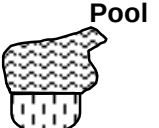
Large woody debris

Overall % cover of logs and branches greater than 10cm in diameter

 _____ % Notes on visibility _____

Extent of bedform features

Total % composition for all features must equal 100%

Height >1m Gradient >60°		Waterfall	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Height (m) _____ Est. Av. Gradient (°)
Step Height <1m Gradient 5-60° Strong currents		Cascade	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Height (m) _____ Est. Av. Gradient (°)
Gradient 3-5° Strong currents Rocks break surface		Rapid	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)
Gradient 1-3° Moderate currents Surface unbroken but unsmooth		Riffle	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)
Gradient 1-3° Small currents Surface unbroken and smooth		Glide	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)
Gradient 1-3° Small but distinct & uniform current Surface unbroken		Run	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)
Area where stream widens or deepens and current declines		Pool	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)
A reasonable sized (>20% of channel width) cut-off section away from		Backwater	_____ % of site _____ Est. Av. Length (m) _____ Est. Av. Depth (m) _____ Est. Av. Width (m)

Note: An additional response variable planform channel pattern is measured in the office

Macrophyte cover Assess % cover of the sampling site by each category.

Overall % cover of macrophytes _____ % cover of emergent macrophytes _____
 % cover of floating macrophytes _____
 % cover of submerged macrophytes _____

Total should equal overall % cover of macrophytes

Macrophyte composition

Use a macrophyte field guide (i.e. Sainty and Jacobs, 1994) to aid identification.

Listed macrophytes can be changed to reflect the common taxa present in each State or Territory. N denotes a native taxa and I denotes an introduced taxa.

Emergent macrophytes

	Present	% cover
<i>Brachiaria</i> (Para Grass) I	☐	_____
<i>Crassula</i> (Crassula) N	☐	_____
<i>Cyperus</i> (Sedge) I/N	☐	_____
<i>Eleocharis</i> (Spikerush) N	☐	_____
<i>Juncus</i> (Rush) I/N	☐	_____
<i>Paspalum</i> (Water Couch) N	☐	_____
<i>Phragmites</i> (Common Reed) N	☐	_____
<i>Ranunculus</i> (Buttercup) I	☐	_____
<i>Scirpus</i> (Clubrush) N	☐	_____
<i>Triglochin</i> (Water Ribbon) N	☐	_____
<i>Typha</i> (Cumbungi) N	☐	_____
Other _____	☐	_____
Other _____	☐	_____
Other _____	☐	_____

Submerged macrophytes

	Present	% cover
<i>Ceratophyllum</i> (Hornwort) N	☐	_____
<i>Chara</i> (Stonewort) N	☐	_____
<i>Elodea</i> (Canadian Pondweed) I	☐	_____
<i>Myriophyllum</i> (Water Milfoil) I/N	☐	_____
<i>Nitella</i> (Stonewort) N	☐	_____
<i>Potamogeton</i> (Pondweed) N	☐	_____
<i>Triglochin</i> (Water Ribbon) N	☐	_____
<i>Vallisneria</i> (Ribbonweed) N	☐	_____
Other _____	☐	_____
Other _____	☐	_____
Other _____	☐	_____

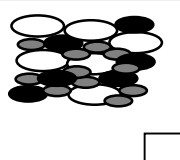
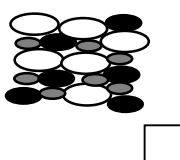
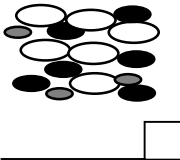
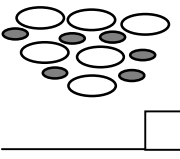
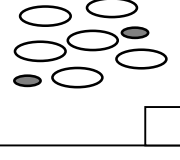
Floating macrophytes

	Present	%
<i>Azolla</i> (Azolla) N	☐	_____
<i>Callitriche</i> (Starwort) I	☐	_____
Other _____	☐	_____
Other _____	☐	_____
Other _____	☐	_____

Overall % cover of native macrophyte taxa _____ } Total should equal overall % cover of macrophytes from above
 Overall % cover of native macrophyte taxa _____ }

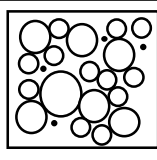
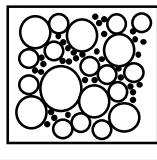
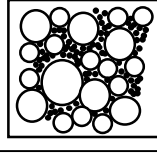
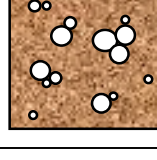
Bed compaction

Choose one category only

	Tightly packed, armoured Array of sediment sizes, overlapping, tightly packed and very hard to dislodge	☐
	Packed, unarmoured Array of sediment sizes, overlapping, tightly packed but can be dislodged with moderate	☐
	Moderate compaction Array of sediment sizes, little overlapping, some packing but can be dislodged with moderate	☐
	Low compaction (1) Limited range of sediment sizes, little overlapping, some packing and structure but can be dislodged very easily	☐
	Low compaction (2) Loose array of fine sediments, no overlapping, no packing and structure and can be dislodged very easily	☐

Sediment matrix

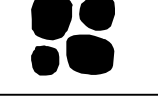
Choose one category only

	Bedrock	☐
	Open framework 0-5% fine sediment, high availability of interstitial spaces	☐
	Matrix filled contact framework 5-32% fine sediment, moderate availability of interstitial spaces	☐
	Framework dilated 32-60% fine sediment, low availability of interstitial spaces	☐
	Matrix dominated >60% fine sediment, interstitial spaces virtually absent	☐

Sediment angularity

Choose one category only

Assess cobble, pebble and gravel fractions only

	Very angular	☐
	Angular	☐
	Sub-angular	☐
	Rounded	☐
	Well rounded	☐
	Cobble, pebble and gravel fractions not present	☐

In the USEPA Habitat Assessment on the following pages, be sure to use the correct form for high or low gradient streams

Bed stability rating Choose one category only

Unstable - eroding

Stable

Unstable - depositing

Severe erosion Streambed scoured of fine sediments. Signs of channel deepening. Bare, severely eroded banks. Erosion heads. Steep streambed caused by erosion.	Moderate erosion Little fine sediment present. Signs of channel deepening. Eroded banks. Streambed deep and narrow. Steep streambed comprised of unconsolidated (loosely arranged and unpacked) material.	Bed stable A range of sediment sizes present in the streambed. Channel is in a 'relatively natural' state (not deepened or infilled). Bed and bar sediments are roughly the same size. Banks stable. Streambed comprised of consolidated (tightly arranged and packed) material.	Moderate deposition Moderate build-up of fine sediments at obstructions and bars. Streambed flat and uniform. Channel wide and shallow.	Severe deposition Extensive build up of fine sediments to form a flat bed. Channel blocked, but wide and shallow. Bars large and covering most of the bed or banks. Streambed comprised of unconsolidated (loosely arranged and unpacked) material.
☐	☐	☐	☐	☐

USEPA Habitat Assessment
Circle a score for each parameter

HIGH GRADIENT STREAMS

Habitat parameter	Condition category																				
	Excellent					Good					Fair					Poor					
1. Epifaunal substrate / available cover	Greater than 70% of substrate favourable for epifaunal colonisation and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonisation potential (i.e. logs/snags that are not new fall and not transient).					40-70% mix of stable habitat; well-suited for full colonisation potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonisation (may rate at high end of scale).					20-40% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.					Less than 20% stable habitat; lack of habitat is obvious; substrate unstable or lacking.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
2. Embeddedness	Gravel, cobble and boulder particles are 0-25% surrounded by fine sediment. Layering of cobble provides diversity of niche space.					Gravel, cobble and boulder particles are 25-50% surrounded by fine sediment.					Gravel, cobble and boulder particles are 50-75% surrounded by fine sediment.					Gravel, cobble and boulder particles are more than 75% surrounded by fine sediment.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
3. Velocity / depth regime	All four velocity/depth regimes present (slow-deep, slow-shallow, fast-deep, fast-shallow). Slow is <0.3m/s, deep is >0.5m).					Only 3 of the 4 regimes present (if fast-shallow is missing, score lower than if missing other regimes).					Only 2 of the 4 habitat regimes present (if fast-shallow or slow-shallow are missing, score low).					Dominated by 1 velocity/depth regime (usually slow-deep).					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
4. Sediment deposition	Little or no enlargement of islands or point bars and less than 5% of the bottom affected by sediment deposition.					Some new increase in bar formation, mostly from gravel, sand or fine sediment; 5-30% of the bottom affected; slight deposition in pools.					Moderate deposition of new gravel, sand or fine sediment on old and new bars; 30-50% of the bottom affected; sediment deposits at obstructions, constrictions and bends; moderate deposition in pools prevalent.					Heavy deposits of fine material, increased bar development; more than 50% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
5. Channel flow status	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.					Water fills >75% of the available channel; or <25% of channel substrate is exposed.					Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.					Very little water in channel and mostly present as standing pools.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
6. Channel alteration	Channelization or dredging absent or minimal; stream with normal pattern.					Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e. dredging (greater than 20 yr) may be present, but recent channelization is not present.					Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted.					Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0

Continued over

USEPA Habitat Assessment
Circle a score for each parameter

HIGH GRADIENT STREAMS

Page 2 of 2

Habitat parameter	Condition category																				
	Excellent					Good					Fair					Poor					
7. Frequency of riffles (or bends)	Occurrence of riffles relatively frequent; ratio of distance between riffles divided by width of the stream <7:1 (generally 5 to 7); variety of habitat is key. In streams where riffles are continuous, placement of boulders or other large, natural obstruction is important.					Occurrence of riffles infrequent; distance between riffles divided by the width of the stream is between 7 to 15.					Occasional riffle or bend; bottom contours provide some habitat; distance between riffles divided by the width of the stream is between 15 to 25.					Generally all flat water or shallow riffles; poor habitat; distance between riffles divided by the width of the stream is a ratio of >25.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
8. Bank stability (score each bank)	Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.					Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.					Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.					Unstable; many eroded areas; 'raw' areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			
9. Vegetative protection (score each bank)	More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understorey shrubs, or non woody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally.					70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not well-represented; disruption evident but not affecting full plant growth potential to any great extent; more than one half of the potential plant stubble height remaining.					50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than one-half of the potential plant stubble height remaining.					Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been removed to 5 centimetres or less in average stubble height.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			
10. Riparian zone score (score each bank)	Width of riparian zone >18 metres; human activities (i.e. roads, lawns, crops etc.) have not impacted the riparian zone.					Width of riparian zone 12-18 metres; human activities have impacted the riparian zone only minimally.					Width of riparian zone 6-12 metres; human activities have impacted the riparian zone a great deal.					Width of riparian zone <6 metres; little or no riparian vegetation is present because of human activities.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			

TOTAL HIGH GRADIENT HABITAT SCORE

Site No. _____ Date _____

USEPA Habitat Assessment
Circle a score for each parameter**LOW GRADIENT STREAMS**

Page 1 of 2

Habitat parameter	Condition category																				
	Excellent					Good					Fair					Poor					
1. Epifaunal substrate / available cover	Greater than 50% of substrate favourable for epifaunal colonisation and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonisation potential (i.e. logs/snags that are not new fall and not transient).					30-50% mix of stable habitat; well-suited for full colonisation potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonisation (may rate at high end of scale).					10-30% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.					Less than 10% stable habitat; lack of habitat is obvious; substrate unstable or lacking.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
2. Pool substrate characterization	Mixture of substrate materials, with gravel and firm sand prevalent; root mats and submerged vegetation common.					Mixture of soft sand, mud or clay; mud may be dominant; some root mats and submerged vegetation present.					All mud or clay or sand bottom; little or no root mat; no submerged vegetation.					Hard-pan clay or bedrock; no root mat or vegetation.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
3. Pool variability	Even mix of large-shallow, large-deep, small-shallow, small-deep pools present.					Majority of pools large-deep; very few shallow.					Shallow pools much more prevalent than deep pools.					Majority of pools small-shallow or pools absent.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
4. Sediment deposition	Little or no enlargement of islands or point bars and less than 20% of the bottom affected by sediment deposition.					Some new increase in bar formation, mostly from gravel, sand or fine sediment; 20-50% of the bottom affected; slight deposition in pools.					Moderate deposition of new gravel, sand or fine sediment on old and new bars; 50-80% of the bottom affected; sediment deposits at obstructions, constrictions and bends; moderate deposition in pools prevalent.					Heavy deposits of fine material, increased bar development; more than 80% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
5. Channel flow status	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.					Water fills >75% of the available channel; or <25% of channel substrate is exposed.					Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.					Very little water in channel and mostly present as standing pools.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
6. Channel alteration	Channelization or dredging absent or minimal; stream with normal pattern.					Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e. dredging (greater than 20 yr) may be present, but recent channelization is not present.					Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted.					Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0

Continued over

USEPA Habitat Assessment
Circle a score for each parameter

LOW GRADIENT STREAMS

Habitat parameter	Condition category																				
	Excellent					Good					Fair					Poor					
7. Channel sinuosity	The bends in the stream increase the stream length 3 to 4 times longer than if it was in a straight line. (Note – channel braiding is considered normal in coastal plains and other low-lying areas. This parameter is not easily rated in these areas).					The bends in the stream increase the stream length 2 to 3 times longer than if it was in a straight line.					The bends in the stream increase the stream 1 to 2 times longer than if it was in a straight line.					Channel straight; waterway has been channelized for a long distance.					
SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
8. Bank stability (score each bank)	Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.					Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.					Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.					Unstable; many eroded areas; 'raw' areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			
9. Vegetative protection (score each bank)	More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understorey shrubs, or non woody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally.					70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not well-represented; disruption evident but not affecting full plant growth potential to any great extent; more than one half of the potential plant stubble height remaining.					50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than one-half of the potential plant stubble height remaining.					Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been removed to 5 centimetres or less in average stubble height.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			
10. Riparian zone score (score each bank)	Width of riparian zone >18 metres; human activities (i.e. roads, lawns, crops etc.) have not impacted the riparian zone.					Width of riparian zone 12-18 metres; human activities have impacted the riparian zone only minimally.					Width of riparian zone 6-12 metres; human activities have impacted the riparian zone a great deal.					Width of riparian zone <6 metres; little or no riparian vegetation is present because of human activities.					
SCORE	Left bank		10	9		8	7	6			5	4	3			2	1	0			
SCORE	Right bank		10	9		8	7	6			5	4	3			2	1	0			

TOTAL LOW GRADIENT HABITAT SCORE

Channel cross-sections and variables to be measured in the area around a cross section

Detailed instructions on the measurement of channel cross-sections are provided in the protocol manual. Be familiar with these before proceeding.

Two cross-sections are required at homogeneous sampling sites (generally lowland streams) and three cross-sections at heterogeneous sampling sites (generally upland streams).

Where the water level at the time of sampling is at or near the water mark level, stream width at the water surface will be equal to stream width at the water mark. In this case, vertical distance between the water surface and the water mark should be entered as 0.

Cross-section sketch

The channel sketch should show in cross-section the shape of the channel and include the location of the water surface, watermark and bankfull points. Also show other features such as bars, rocky outcrops and snags encountered at the cross section.

Cross-section number _____ of _____

Type of bedform at the cross-section
☐ Riffle ☐ Run ☐ Pool ☐ Cascade ☐ Other _____

Bankfull channel width (m) _____
 (=total of boxes A+B+C)

Stream width at the water mark (m) A

Stream width at the water surface (m)

Bank height (m)

Bank width (m) B

Vertical distance between the water surface and the water mark (m)

Horizontal distances (m)

Vertical water depths (cm)

Bank height (m)

Bank width (m) c

Vertical distance between the water surface and the water mark (m)

Notes on cross-section measurement**Riparian zone width**

Left bank _____ (m) Right bank _____ (m)

Bank material Assess % composition for each bank

	Left bank	Right bank
Bedrock	_____	_____
Boulder (>256mm)	_____	_____
Cobble (64-256mm)	_____	_____
Pebble (16-64mm)	_____	_____
Gravel (2-16mm)	_____	_____
Sand (0.06-2mm)	_____	_____
Fines (silt and clay, <0.06mm)	_____	_____
	} Total 100% each	

Substrate composition

Assess % composition in the area of bed 5m either side of the cross-section.

Bedrock	_____
Boulder (>256mm)	_____
Cobble (64-256mm)	_____
Pebble (16-64mm)	_____
Gravel (2-16mm)	_____
Sand (0.06-2mm)	_____
Fines (silt and clay <0.06mm)	_____

Total 100%

Filamentous algae cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Periphyton cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Moss cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Detritus cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Assess in the area 5m either side of the cross section

Channel cross-sections and variables to be measured in the area around a cross section

Detailed instructions on the measurement of channel cross-sections are provided in the protocol manual. Be familiar with these before proceeding.

Two cross-sections are required at homogeneous sampling sites (generally lowland streams) and three cross-sections at heterogeneous sampling sites (generally upland streams).

Where the water level at the time of sampling is at or near the water mark level, stream width at the water surface will be equal to stream width at the water mark. In this case, vertical distance between the water surface and the water mark should be entered as 0.

Cross-section sketch

The channel sketch should show in cross-section the shape of the channel and include the location of the water surface, watermark and bankfull points. Also show other features such as bars, rocky outcrops and snags encountered at the cross section.

Cross-section number _____ of _____

Type of bedform at the cross-section
☐ Riffle ☐ Run ☐ Pool ☐ Cascade ☐ Other _____

Bankfull channel width (m) _____
 (=total of boxes A+B+C)

Stream width at the water mark (m) A

Stream width at the water surface (m)

Bank height (m)

Bank width (m) B

Vertical distance between the water surface and the water mark (m)

Horizontal distances (m)

Vertical water depths (cm)

Bank height (m)

Bank width (m) C

Vertical distance between the water surface and the water mark (m)

Notes on cross-section measurement**Riparian zone width**

Left bank _____ (m) Right bank _____ (m)

Bank material Assess % composition for each bank

	Left bank	Right bank
Bedrock	_____	_____
Boulder (>256mm)	_____	_____
Cobble (64-256mm)	_____	_____
Pebble (16-64mm)	_____	_____
Gravel (2-16mm)	_____	_____
Sand (0.06-2mm)	_____	_____
Fines (silt and clay, <0.06mm)	_____	_____
	} Total 100% each	

Substrate composition

Assess % composition in the area of bed 5m either side of the cross-section.

Bedrock	_____
Boulder (>256mm)	_____
Cobble (64-256mm)	_____
Pebble (16-64mm)	_____
Gravel (2-16mm)	_____
Sand (0.06-2mm)	_____
Fines (silt and clay <0.06mm)	_____
	} Total 100%

Filamentous algae cover

Assess in the area 5m either side of the cross section

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Periphyton cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Moss cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Detritus cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

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Type of bedform at the cross-section
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Bank height (m)

Bank width (m) B

Vertical distance between the water surface and the water mark (m)

Horizontal distances (m)

Vertical water depths (cm)

Bank height (m)

Bank width (m) C

Vertical distance between the water surface and the water mark (m)

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Gravel (2-16mm)	_____	_____
Sand (0.06-2mm)	_____	_____
Fines (silt and clay, <0.06mm)	_____	_____
	} Total 100% each	

Substrate composition

Assess % composition in the area of bed 5m either side of the cross-section.

Bedrock	_____
Boulder (>256mm)	_____
Cobble (64-256mm)	_____
Pebble (16-64mm)	_____
Gravel (2-16mm)	_____
Sand (0.06-2mm)	_____
Fines (silt and clay <0.06mm)	_____
	} Total 100%

Filamentous algae cover

Assess in the area 5m either side of the cross section

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Periphyton cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%

Moss cover

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Detritus cover

☐ <10% ☐ 10-35% ☐ 35-65% ☐ 65-90% ☐ >90%