VICTORIAN NATIONAL PARKS ASSOCIATION

NATURE CONSERVATION REVIEW

FRESHWATER DEPENDENT ECOSYSTEMS

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1 SCOPE & SIGNIFICANCE OF FRESHWATER BIODIVERSITY IN VICTORIA

Victoria's natural ecosystems support at least 3,140 native species of vascular plants, 900 lichens, 750 mosses and liverworts, 111 mammals, 447 birds, 46 freshwater and 600 marine fish, 133 reptiles, 33 amphibians, and an untold number of invertebrates, fungi and algae (DSE 2003).

Victoria has a greater density of rivers and streams than any other mainland State. There are 3,820 named watercourses in Victoria, with a total length of 56,000 km. In addition, there are numerous tributaries (and sometimes, distributaries) associated with these named watercourses.

Victoria has 17 river segments/corridors with Heritage River status (LCC 1991). They are essentially river reaches judged to have at least one value of national or international significance and at least four values of State of greater significance. The types of values considered span a wide spectrum and include biological attributes (e.g. botanical and faunal qualities, diversity of native fish species and presence of endangered or vulnerable fish species), environmental attributes (e.g. geological and geomorphological features), scenic qualities, cultural heritage and recreational qualities (LCC 1991). The 17 Heritage River corridors occur along the Mitta Mitta, Ovens, Howqua, Big, Goulburn, Wimmera, Genoa, Bemm (and its tributaries, Goolengook, Arte, and Errinundra), Snowy, Suggan Buggan and Berrima, Upper Buchan, Mitchell and Wonnangatta, Thomson, Yarra, Lederderg, Aire, and Glenelg Rivers (LCC 1991).

Victoria has approximately 16,700 non-flowing wetlands covering 540,900 hectares, of which 12,800 (covering 432,800 hectares) are natural and the remaining 3,900 wetlands are artificial (DSE 2007b). Victoria also has 11 internationally important wetland systems that have been listed as Ramsar sites under the Convention on Wetlands. 10 of these were listed in 1982 and include: Corner Inlet, Gippsland Lakes, Barmah Forest, Gunbower Forest, Hattah-Kulkyne Lakes, Kerang Wetlands, Lake Albacutya, Port Phillip Bay (Western Shoreline) and Bellarine Peninsula, Western District Lakes, and Western Port. Victoria's latest Ramsar site is the Edithvale-Seaford Wetlands, in southeast metropolitan Melbourne, and was listed in August 2001. Victoria also has 159 wetlands of national importance.

These riverine ecosystems provide a rich diversity of habitats that support two species of freshwater mammals (the platypus *Ornithorhynchus anatinus* and water rat *Hydromys chrysogaster*, over 100 species of waterbirds, 33 species of amphibians, 46 species of freshwater fish and an undetermined number of invertebrate species (CES 2008).

High levels of endemism have been identified but uncertainty remains over the current and reference distributions of many of Victoria's aquatic fauna. Notwithstanding this uncertainty, a total of 21 freshwater and estuarine fish species are listed as threatened under the *Flora and Fauna Guarantee Act 1988* (hereafter referred to as the *FFG*) (DSE 2009b). Three species of native fish (Agassiz's chanda perch *Ambassis agassizii*, freshwater herring *Potamalosa richmondia* and southern purple spotted gudgeon *Mogurnda adspersa*) are considered regionally extinct under the Advisory List of Threatened Vertebrate Fauna in Victoria (DSE 2007a). Seven species are considered critically endangered, namely, trout cod *Maccullochella macquariensis*, silver perch *Bidyanus bidyanus*, barred galaxias *Galaxias fuscus*, river blackfish (upper Wannon River form) *Gadopsis marmoratus*, Murray hardyhead *Craterocephalus fluviatilis*, Australian mudfish *Neochanna cleaveri* and Australian (Tasmanian) whitebait *Lovettia sealii*. Five species are considered endangered, including freshwater catfish *Tandanus tandanus*, Macquarie perch

Macquaria australasica, Murray cod *Maccullochella peelii peelii*, variegated pygmy perch *Nannoperca variegate* and Cox's gudgeon *Gobiomorphus coxii* . A further six freshwater species were considered vulnerable (DSE 2007a).

Frog populations throughout Victoria have declined and 11 out of a total 33 species are considered threatened under the *FFG* (DSE 2009b). Two species of turtle are considered threatened under the *FFG*; the leathery turtle *Dermochelys coriacea* is considered to be critically endangered and the broad-shelled turtle *Chelodina expansa* is considered endangered (DSE 2007a).

78 species of birds are listed as threatened under the *FFG* (DSE 2009b). Of these, 31 species generally inhabit freshwater and/or coastal and marine environments, although many more of the listed bird species inhabit water-dependent ecosystems such as floodplain forests and woodlands (see below).

Freshwater macro-invertebrates are a diverse group of insects, crustaceans and molluscs that include shrimp, crayfish, mussels, snails, water boatmen, dragonflies, stoneflies and worms. The number of macro-invertebrate species in Victorian freshwater systems is unknown but is estimated to greatly exceed the diversity of vertebrate fauna. 18 crustacean species are listed as threatened in Victoria (DSE 2009b).

This report discusses scientific approaches to freshwater biodiversity conservation and their applicability in Victoria. The usage of the term "freshwater biodiversity" in this report encompasses the biota of both surface water-dependent ecosystems (SDEs) and groundwater-dependent ecosystems (GDEs). Surface water-dependent ecosystems refers to all lotic and lentic ecosystems that depend of flowing or still surface waters respectively (e.g. rivers, streams and springs as well as pools, lakes, ponds and swamps). Groundwater-dependent ecosystems are of three main categories (after Eamus *et al*. 2006):

a) reliant on surface expression of groundwater;
b) reliant on subsurface groundwater accessible within the rooting depth of aboveground vegetation; and
c) reliant on subsurface groundwater within wholly subterranean aquifer and cave systems.

Rivers and streams with perennial flow and permanent wetlands in a floodplain system are often examples of GDEs reliant on the surface expression of groundwater, as they are likely to be deriving a significant portion of their baseflow or freshwater input from groundwater discharge. Examples of the second type of GDE include River Red Gum Forests (such as those along the lower River Murray) and paperbark swamp forests. The third type of GDE is little known and appreciated, being largely 'invisible'.

As Tomlinson & Boulton (2008) concluded from their overview of the biodiversity of subsurface groundwater ecosystems, there are extensive gaps in our knowledge of the distribution, composition and biodiversity value of Australian stygofauna (groundwater animals). Despite this incomplete inventory, stygofauna are present across a variety of Australian subsurface environments and are generally characterized by high diversity and local-scale endemicity (Boulton *et al*. 2003). Groundwater ecosystems are relatively stable environments (compared to surface water environments) and may be very persistent through geological time through major episodes of climate change, ice ages, tectonic and orogenic events. This means some aquifers may be "living museums containing a sample of the lineages that comprised the faunas from various geological periods" (Humphreys 2009) and are great scientific interest.

Finally, I note that there are estuarine and costal ecosystems such as mangroves, salt marshes and sea grass beds which qualify as GDEs because they rely on the submarine discharge of groundwater (Tomlinson & Boulton 2008), but these are beyond the scope of the present report will not be further considered.

2 Freshwater-dependent Ecosystems: Ecological Characteristics, Functions, Processes & Interactions

There is a long history of work attempting to conceptualize and theorise our understanding of the complex form and functioning of river ecosystems. Prominent models include the River Continuum Concept (RCC) (Vannote *et al*. 1980), the Flood Pulse Concept (FPC) (Junk *et al*. 1989; Tockner *et al*. 2000), the Riverine Productivity Model (RPM) (Thorp & Delong 1994) and the River Ecosystem Synthesis (RES) (Thorp *et al*. 2006). These models (and their variants) have been proposed to explain patterns of structural and functional biocomplexity across spatiotemporal scales in river networks. All are heuristic and aimed at facilitating scientific exploration.

It is not the object of this section to provide detailed accounts of these various models. Rather, the aim here is to describe key ecological characteristics of freshwater environments, their ecological functions, processes and interactions to set the context for understanding issues that ought to be taken into account when considering freshwater protection and conservation prioritization (§ 3).

2.1 Ecological Characteristics & Processes

Climate (which influences runoff, flow characteristics, riparian, floodplain and aquatic vegetation) and catchment geomorphology (which influences channel geometry, i.e. width, depth, slope, meander wavelength, sinuosity and width-depth ratio) are important determinants of landscape-scale riverine structure, patterns and processes (Thoms 2006; Thorp *et al*. 2006). Additional complexity is overlaid by the effect of upstream and downstream features along a flow path (such as dams, weirs, waterfalls and logjams) and natural disturbances (e.g. fire, landslides) and anthropogenic activities in the upstream contributing catchment area.

Freshwater ecosystems are incredibly diverse, heterogeneous and complex because they are driven by variability in four dimensions. Rivers have interactive pathways along three spatial dimensions (Ward 1989): longitudinal (headwater-estuarine), lateral (river channel-riparian-floodplain), and vertical (river channel, riparian floodplain-groundwater). The fourth dimension, time, superimposes a temporal hierarchy on the three spatial dimensions (Ward 1989). Although biophysical conditions and biotic communities may change in a continuous, predictable pattern along the longitudinal dimension in some river networks (e.g. from headwaters to medium-size rivers), discontinuous patterns along longitudinal and lateral dimensions of Victorian river networks are also common, particularly in lowland, dryland and floodplain river networks.

The natural flow regime refers to a river system's characteristic temporal patterns of flow magnitude, frequency, duration and predictability. For ease of reference and communication, the flow regime is often decomposed into a number of flow components and given descriptive labels, for example, 'summer-autumn low flows', 'spring freshes', 'bankfull flows', 'overbank flows' and 'cease-to-flow'. The natural flow regime is of profound importance in the structuring and functioning of riverine ecosystems and shaping the life history strategies of freshwater-dependent biota (Poff *et al*. 1997; Bunn & Arthington 2002; Lytle & Poff 2004). At site-scales, interactions of the flow regime with channel bed and banks as well as in-channel geomorphologic features such as pools, runs, bars, benches, overhanging banks and anabranches as well as structural elements such as sediment, pebbles, boulders, tree roots, coarse woody debris and macrophytes creates complex, biophysically heterogeneous habitats. At finer scales still, these interactions produce flow patterns such as slackwaters, eddies, transverse flows and velocity gradients which constitute diverse hydraulic environments for aquatic biota (Crowder and Diplas 2006).

The dynamic nature of the flow regime interacting with the geomorphic template results in spatially and temporally variable habitat patches which provide riverine biota with habitat for attachment, primary production, feeding, resting, refuge, breeding and rearing. Each flow component facilitates a range of riverine functions and processes. For instance, freshes can create new habitat patches through inundation where none existed previously and can alter the nature of a habitat patch from a pool to a run. Freshes help maintain or improve water quality in pools by providing an input of fresh water and mixing or flushing pools which hay have stagnated and/or stratified. Freshes may also establish temporary longitudinal and/or lateral connectivity between different habitat patches thereby transporting or enabling the movement of organic or inorganic material such as sediment, propagules and organisms (Chee *et al*. 2006). The ecological roles/functions associated with the various flow components were reviewed in the process of developing the Victorian Environmental Flows Monitoring and Assessment Program (VEFMAP, see Chee *et al*. 2006).

The lateral dimension of river networks includes the riparian zone of the main channel, sub-bankfull indundation areas, backwaters, low-lying flood runners, anabranches and floodplain wetlands. These features may be wetted, activated or filled by flow components such as freshes, winter-spring high flows, bankfull flows and overbank flows. When this occurs, the soil moisture in riparian and floodplain zones is replenished thereby helping to maintain *in situ* vegetation communities. The moisture gradient extending laterally from the river channel maintains the pattern of zonation which is characteristic of the riparian zone community structure and diversity. Additional (temporary/ephemeral) habitats may be created, providing an assortment of habitat patches of variable quality and spatial arrangement in the floodplain. The slackwater habitats thus created provide refuge from flow currents (which confers energetic advantages) and are often highly productive environments in spring-early summer. They therefore provide suitable hatching, rearing, feeding and refuge environments for riverine biota such as zooplankton and the young stages of shrimp and fish.

Through connectivity in the vertical dimension, streams interact with groundwater in many types of landscapes. At the river scale, stream gains/losses (or alternatively, if we take groundwater as the reference, groundwater recharge/discharge) depends upon factors such as climate, landscape position, geomorphology and geology which affect spatial variation in depth to groundwater and sediment permeability. Interaction patterns for a generic longitudinal sequence displaying typical combinations of factors are outlined in Table 1 (after MDBA 2009).

Table 1. Generic patterns of surfacewater-groundwater interactions as a function of landscape position, depth to groundwater and vertical connectivity.

Landscape position	Depth to groundwater	Connected	Interaction
Steep, upland	shallow	yes	stream gains with groundwater discharge to stream bed

Narrow, alluvial valleys	shallow	yes	flux in both directions, but net outflow from stream bed and banks recharges groundwater
Wide alluvial plains (semi-arid/arid)	deep	no	stream disconnected/separated from groundwater system by an unsaturated zone underlying stream. Stream bed loses water at a rate proportional to permeability of the unsaturated strata
Wide alluvial plain with fine alluvial materials	shallow	yes	flux in both directions, but stream bed gains with net groundwater discharge to stream. (Although this may vary during periods of flooding.)

At finer spatial scales such as the site-scale, spatial variations in stream bed topography, changes in flow direction induced by the presence of meanders, flow obstacles (e.g. boulders, debris pile-ups, macrophyte stands) and geomorphic features (e.g. riffles, sand and gravel bars) as well as sediment permeability generate a mosaic of patches of surface-groundwater exchanges (Brunke & Gonser 1997; Malard *et al*. 2002; Boulton 2007). Surface water downwells (infiltrates) into the sediments and travels along the subterranean flow path through saturated sediments, mixing with groundwater before upwelling into the stream again (Malard *et al*. 2002). The saturated interstitial spaces in the stream bed and bank where surface water and groundwater mix is known as the hyporheic zone. As downwelling water travels through the hyporheic zone mixing with groundwater, its chemical composition is altered by biogeochemical processes typically mediated by microbial biofilms on sediment particles. According to Boulton (2007), the chemically transformed water emanating from the upwelling zones can promote growth of periphyton at the stream bed surface, creating localised 'hotspots' of productivity.

Hyporheic zones represent a spatially and temporally fluctuating ecotone where substrate particle size, interstitial pore space and gradients of temperature, oxygen and nutrients generated by surface-subsurface exchanges create diverse habitat patches. This in turn governs the distribution and abundance of microorganisms, meiofauna, and macroinvertebrates within the sediments as well as the types and rates of metabolic activity and biogeochemical processes (Boulton *et al*. 1998; Malard *et al*. 2002).

Acting in concert, bioclimatic, hydrologic and geomorphic processes create complex mosaics of habitat patches at multiple spatiotemporal scales. In natural settings, the quantity, quality, physical properties and spatial arrangement of habitat types will determine the type (composition) and abundance of the biotic community as well as the rates of ecological processes. In human-modified landscapes, water resource and land use management actions impose additional influences on riverine characteristics and processes.

Riverine ecosystems are particularly valuable because of their remarkable characteristic of creating ecotones at multiple scales. At catchment and landscape scales, riverine ecosystems provide mesic environments in what might otherwise be semi-arid/arid regions. Good examples of this in Victoria include the lower Wimmera and Murray Rivers. At the river scale, the moisture gradient extending laterally creates terrestrial ecotones in the riparian zone and floodplain. The moisture gradient extending vertically and laterally creates subterranean ecotones (the hyporheic zone described above). At the river site or patch scale, ecotones occur where fluctuating flows interact with higher-elevation features within the channel such as bars, benches. At ecotones, the juxtaposition of contrasting environments provides not only a greater diversity but qualitatively different habitats, which in turn supports a greater number and diversity of species.

2.1.1 Ecological functions & processes

Ecosystem functions are the physical, chemical and biological exchanges and processes that contribute to the self-maintenance and self-renewal of an ecosystem. The term 'ecosystem function' has been subject to various and sometimes contradictory interpretations – sometimes being applied to describe the internal functioning of the ecosystem (e.g. nutrient and energy cycling, food-web interactions etc.) and sometimes to describe benefits derived by humans from the properties and processes of ecosystem (e.g. forage production, waste treatment, recharge of aquifers that store and supply water for human use etc) (de Groot *et al*. 2002). In this report, 'ecosystem function' is used strictly in the former sense. The term 'ecosystem services' is used to describe the myriad conditions and processes through which natural ecosystems and the species they contain, sustain and fulfill human life (Daily 1997). In the context of riverine systems, this would include goods such as water for human, livestock and irrigation use, services such as water filtration and purification, a conduit for transport, flood attenuation, maintenance/renewal of floodplain fertility via alluvial deposition, provision of recreational opportunities and intangible aesthetic and cultural benefits.

As Ricklefs *et al*. (1984) elaborates, "Ecological processes include all the physical processes and the plant and animal activities which influence the state of ecosystems and contribute to the maintenance of their integrity and genetic diversity, and thereby their evolutionary potential. The particular processes that make up the dynamics of an individual ecosystem are so numerous and their expressions so diverse that they defy simple characterization. They must be defined individually in each situation. Ecological processes may be most easily appreciated in terms of their results, as identified by the movement of energy, materials, and nutrients; by the information which regulates these functions; and by community changes following disturbance."

Broad categories of ecological processes of direct relevance to freshwater ecosystems include: a) regulation (of the hydrological cycle and biogeochemical cycles, i.e. storage, transport and transformation of water, minerals and organic matter); b) primary production and secondary production, i.e. capture, transformation and flow of energy through foodwebs; c) formation and maintenance of biophysical habitats; d) movement and transport of the various life-history stages of microorganisms, plants and animals; e) biological interactions (e.g. competition, herbivory, predation and pollination) and f) natural disturbance regimes (Ricklefs *et al*. 1984; McGregor *et al*. 2008).

These ecological processes are mediated by the interplay between climate, the flow regime, the geomorphological template, groundwater systems, (instream, riparian, floodplain and catchment) vegetation and the activities of animals. Figure 1 depicts a simplified conceptual model of the main hydrologic, geomorphic and ecological interactions and processes in riverine ecosystems. The previous section has already touched on a range of ecological functions and processes particularly in relation to the creation, provision and maintenance of habitat quantity, quality and diversity. Below, I provide a few illustrations of the other categories of ecological processes.

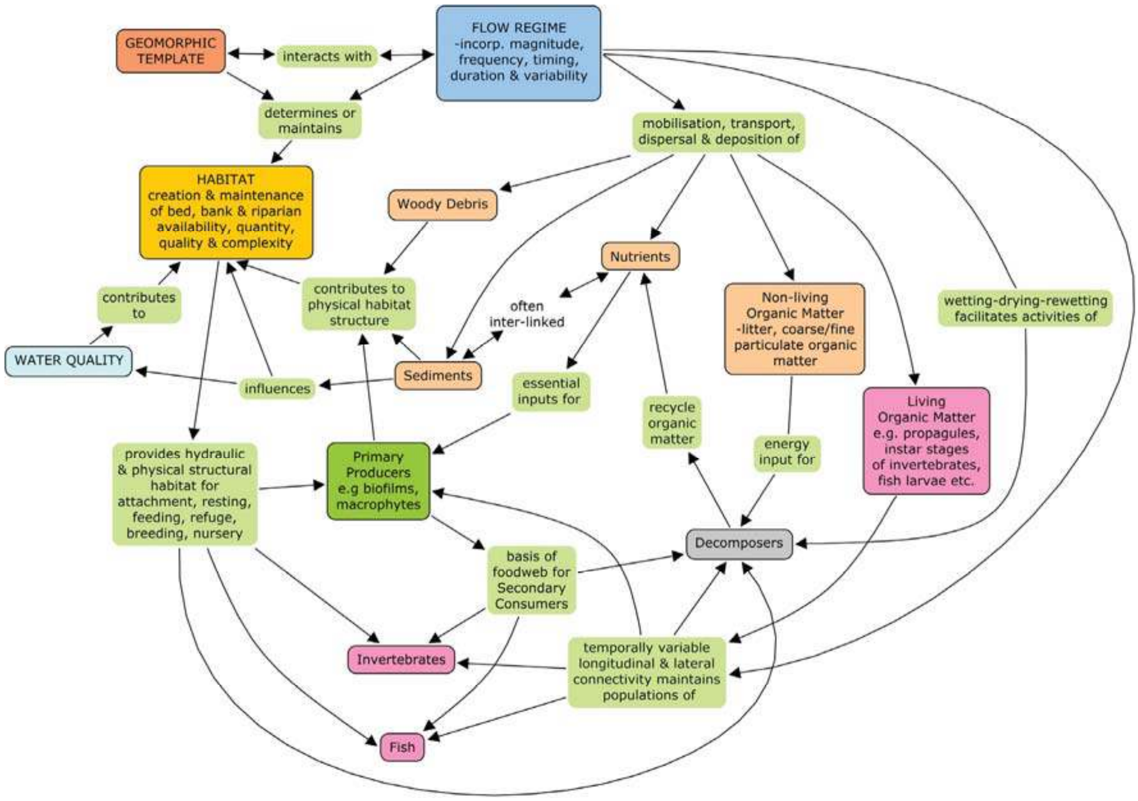


Figure 1. Simplified conceptual model of hydrologic, geomorphic and ecological interactions and processes in riverine ecosystems.

As receptors, storages, and transmitters of water, groundwater systems (aquifers) regulate parts of the hydrological cycle, absorbing runoff and stream flows through river channels as well as the floodplain. This process provides a moderating (buffering) effect on rates of flow changes during periods of flooding. When flooding recedes, aquifer storages release water back to the stream, sustaining flow rates and again buffering rates of flow and river level changes. This phenomenon has implications for riverine biota because changes in rates of flow and water levels affect water velocity, shear stress, intensity of scouring disturbance etc with attendant impacts on biota such as uprooting of seedling and adult macrophytes. Rapid recession of flood flows may strand biota in floodplain environments that not suitable for longer-term survival. From the perspective of humans, the mitigation of flood magnitude and rapid flow rate changes constitutes an ecosystem service which we call 'flood attenuation'.

Habitat patches provide riverine biota with the environments for attachment, primary production, feeding, resting, refuge and generally completing various life-history stages. Habitat patches have a spatial relationship to each other and the integrity of ecosystem function depends on the transport of materials into and out of areas. This includes the longitudinal, lateral and vertical

a) transport of nutrients (often bound up with sediments) required for primary production by autotrophs such as periphyton, phytoplankton, biofilms, submerged, amphibious and riparian vegetation;
b) transport of allochthonous sources of organic material for secondary consumers and decomposers;
c) movement of different life-history stages of microorganisms, plants and animals to allow dispersal to habitats appropriate for completing a specific life-history stage or dispersal to favourable environments with reduced competition for resources or to recolonize habitat patches where local extinction has occurred

The movement of organic (e.g. detritus) and inorganic (e.g. sediment and nutrients) materials by fluvial means depends upon the nature (i.e. extent, duration, continuity etc) of longitudinal, lateral and vertical hydrologic connectivity. In the case of organisms, these movements between habitat patches occur as individuals search for food, defend breeding territories, migrate according to life-history requirements, and disperse from areas of high population concentrations to less densely populated areas. They are critical to the regulation of local populations, determine the area required to avoid extinction, reduce inbreeding within populations, and preserve normal social organization (Ricklefs *et al*. 1984).

The interplay of geomorphological features and the flow regime plays an important role in biogeochemical cycling and energy transfers through the riverine system. During periods of flow cessation or low flows, terrestrial organic matter contributed by the (lateral) riparian zone accumulates and dries in higher-elevation features within the channel such as bars and benches. As flow levels fluctuate, the accumulated organic matter is subjected to cycles of wetting-drying and rewetting. This facilitates physical breakdown as well as decomposition of the organic matter by microbial activity (Baldwin & Mitchell 2000). The resultant dissolved and fine particulate organic matter can be used by microbes, zooplankton, algae, macrophytes and microfauna. Flows then entrain this fresh pool of nutrients and carbon inputs and redistributes them throughout the river system thereby contributing to biogeochemical cycling. Hyporheic zones can also serve as highly reactive sites for nutrient cycling with microbes playing an important functional role in mineralizing organic matter, reducing nitrate loading and regulating the export of nutrients back to surface waters (Pinay *et al*. 2002; Tomlinson & Boulton 2008). From the perspective of humans, these processes provide the ecosystem services of nutrient cycling and maintenance of water quality.

Most ecological systems undergo cycles of disturbance and recovery which occur on characteristic scales of space and time. Disturbance may be necessary to maintain local ecological processes and can take many forms with smaller scale disturbances generally occurring more frequently than larger ones. Examples of small scale disturbances include intra-annual cease-to-flow periods or freshes/bankfull flows with the capacity do geomorphic work such as flushing sediment from the channel bed substrate or scouring the channel bed and banks and redistributing sediment to produce geomorphic features which increase channel complexity. An example of a large scale disturbance might be a large inter-annual overbank flooding event.

Headwater sections and portions of a river which traverses semi-arid/arid landscapes typically experience periods of flow cessation. During such periods, the river may contract to a series of isolated pools, or portions of the channel may dry out. This produces a range of effects, for instance: (a) biota in these pools are likely to be subjected to stresses such as intensified predation, competition and physicochemical stresses (e.g. low dissolved oxygen concentrations) and (b) exposure of large areas of the streambed may act as a desiccation disturbance mechanism which resets successional processes for macroinvertebrate and vegetation communities.

It is believed that desiccation disturbance prevents the system from being dominated by any particular group of organisms. Biota in dryland river systems have special physiological or behavioural adaptations that allow them to persist in harsh conditions in locations which they might otherwise be displaced by dominant but less tolerant species. In the short-term there may be localised extinction of certain species. And in the long-term, changes in diversity and biomass. Natural recovery of the system may depend upon recolonization from outside the area or the availability of effective refuges for local populations during periods of flow cessation.

Disturbances themselves might be vital for the maintenance of the integrity and diversity of the ecosystem. Without such disturbances, the system may tend towards geographical and temporal uniformity, with a resulting reduction in the variety of life forms and important plant and animal species (Ricklefs *et al*. 1984). Flow variability is an inherent feature of the natural flow regime of many Australian rivers, particularly ones which flow for part of their length through semi-arid/arid landscapes (Walker *et al*. 1995; Puckridge *et al*. 1998; Bunn *et al*. 2006). Flow variability can function as a form of disturbance. For example, variation in water levels is an important driver of lateral zonation patterns of aquatic, amphibious and riparian vegetation communities and may help maintain species diversity in the emergent and marginal aquatic vegetation communities. When water levels are stable for a prolonged period the plants that establish and persist close to the water line tend to be species more associated with lentic (wetland) environments than lotic (flowing water) environments. Absence of flow variability may also result in wider zones of terrestrial or flood intolerant plant species and a shrinking in the width of the zone characterized by flood tolerant species (Chee *et al*. 2006).

2.2 Threatening processes

Auld & Keith (2009) proposed a simple threat classification of 5 major groupings: habitat loss and degradation; change to natural disturbance regimes; dysfunction of biological interactions; over-exploitation and climate change. They note that while interactions and synergies between different threatening processes inevitably blur the distinctions between these broad classes, the categories nevertheless provide a useful framework for conceptualizing threatening processes and assessing policy and management approaches.

The latest (July 2009) update of Victoria's Flora and Fauna Guarantee Act 1988 lists 38 potentially threatening processes (DSE 2009a). 12 of the 38 are directly relevant to (non-marine, non-estuarine) freshwater biodiversity. Table 2 summarizes these threatening processes according to Auld & Keith's (2009) classification scheme (with some fitting into more than one threat grouping). The *FFG*-listed threats are comparable and consistent with those that have been identified for freshwater biodiversity from global perspectives (see Pringle 2001; Abell *et al*. 2007).

Table 2. *Flora and Fauna Guarantee Act*-listed potentially threatening processes of direct relevance to freshwater biodiversity, organized according to Auld & Keith's (2009) threat classification scheme.

Threat grouping	*FFG*-listed potentially threatening processes
Habitat loss & degradation	• Alteration to the natural flow regime • Alteration to the natural temperature regimes of rivers and streams • Degradation of native riparian vegetation along Victorian rivers and streams • Increase in sediment input into Victorian rivers and streams due to human activities • Input of toxic substances into Victorian rivers and streams • Removal of wood debris from Victorian streams • Loss of coarse woody debris from Victorian native forests and woodlands • Loss of hollow-bearing trees from Victorian native forests • Wetland loss and degradation as a result of change in water regime, dredging, draining, filling and grazing
Changes to natural disturbance regimes	• Alteration to the natural flow regime • Alteration to the natural temperature regimes of rivers and

Dysfunction of biological interactions (e.g. competition, herbivory, predation, pathogens, pollination etc)	• Infection of amphibians with Chytrid Fungus, resulting in chytridiomycosis • *Introduction of live fish into waters outside their natural range within a Victorian river catchment after 1770 • Invasion of native vegetation by 'environmental weeds' • Prevention of passage of aquatic biota as a result of the presence of instream structures
Over-exploitation	• Alteration to the natural flow regime
Climate Change	-

* An interesting conundrum is the freshwater catfish, an *FFG*-listed species which was introduced into the Wimmera River in the 1970s. The freshwater catfish is not endemic to the Wimmera River catchment and has declined in distribution and abundance throughout its natural range in Victoria. However, it has established self-sustaining populations in the Wimmera River and thrives to the extent that the Wimmera River and its tributaries are the only waters in Victoria where the species may be legally caught.

In Australia, the effects of habitat loss and degradation operate at a range of spatial scales but are most prevalent and intense in landscape settings that are suitable for agricultural production, natural resource extraction and urban development. Direct pressures from water resource development and efforts to ensure security of supply in our highly-variable hydrological systems include dam construction, water extraction and flow regulation, stream channelization and desnagging, the draining of wetlands and construction of levees. Indirect pressures include native vegetation clearing in catchments, agricultural development and attendant effects of erosion, sedimentation, nutrient run-off and alien species introduction (CES 2008). Human activities that disrupt the hydrological cycle, geomorphic and ecological processes often produce a cascade of effects on riverine and terrestrial ecosystems. The hydrological and functional connections between surface water and groundwater mean that threatening processes that affect one is likely to have a flow on effect on the other (Tomlinson & Boulton 2008).

Common manifestations of the alteration of the natural flow regime in Victorian river systems include: a) loss of flow variability, b) extended periods of cease-to-flow or low flow, c) reduced flood frequencies and flood magnitudes, d) reversal of flow seasonality (i.e. unseasonal high flows during the summer-autumn low flow period and conversely, unseasonal low flows during the winter-spring high flow period) and e) loss of cease-to-flow periods. Many river systems, particularly ones subject to some form of regulation will experience more than one form of alteration to its natural flow regime.

An important property of threatening processes is that they operate in ways that affect multiple species and ecosystem processes more or less simultaneously (Auld & Keith 2009). This is particularly true in freshwater ecosystems because of the primacy of the natural flow regime which shapes the geomorphic template and underpins the fluxes and movements which link together the ecological processes occurring in different places and in different habitats. Reducing and interfering with habitat processes and ecological processes associated with the movement and exchange of organisms inevitably leads to marked change in habitat, population and ecosystem structure (Ricklefs *et al*. 1984). However, the primacy of the natural flow regime means that actions to address the threat of alteration to the natural flow regime should produce broad benefits in the form of reduced risks to multiple species and processes. The provision and management of environmental flows to ameliorate, rehabilitate or restore degraded freshwater ecosystems is a relatively new endeavour with many complex challenges at the design, implementation and monitoring and assessment stages (Chee *et al*. 2006; Webb *et al*., *in review*). Environmental flows, on their own however, cannot address threatening processes such as increased input of sediment, pollutants and contaminants, removal of woody debris and dysfunction of biological interactions caused by the introduction of alien species.

Potentially threatening processes to groundwater-dependent ecosystems fall mainly into the categories of habitat loss and degradation and over-exploitation. For instance, alteration of the natural flow regime and changes in land use (e.g. removal of native vegetation cover in the catchment, grazing and accelerated erosion of riparian zone and application of fertilizers) have the potential to increase sediment input and inputs of nitrates, phosphates and toxic substances which can result in changes to groundwater systems. Increased inputs of fine sediments can clog the top layer of channel sediments, reducing pore volume, consolidating the sediment matrix and decreasing permeability of the stream bed (Brunke & Gonser 1997). This can threaten the habitat and consequently, distribution and abundance of microorganisms, meiofauna, and macroinvertebrates within the sediments. Hindrance of exchange processes between surface water and groundwater and over-loading of pollutants and contaminants can also affect the biodiversity and the nature and rates of biogeochemical processing of the hyporheic zone (Boulton *et al*. 2003; Tomlinson & Boulton 2008).

In Victoria, groundwater provides drinking water for approximately 80 cities and towns including Geelong, Ballarat, Portland and Sale. It is also used to irrigate crops, supply drinking water for stock and for industrial purposes (DSE 2009c). Like surface water, groundwater is allocated for commercial and irrigation uses under licensing arrangements under the *Water Act 1989*.

Groundwaters are only recharged when surface waters seep into aquifers. Therefore, over-exploitation or the extraction of groundwater at rates exceeding recharge depletes aquifers. Human activities that disrupt the hydrological cycle in ways that change the quantity and quality of recharge (e.g. reduction in flood frequencies and magnitudes) also impact on groundwater levels, flow and quality (Tomlinson & Boulton 2008). Loss of storage volume and lowering of water levels through overextraction necessarily reduces habitat for stygofauna (Tomlinson & Boulton 2008). Diminished contributions to river baseflows and permanent wetlands has repercussions for biodiversity and ecological functions and processes (Boulton & Hancock 2006). The lowering of water levels below the accessible rooting depth of aboveground terrestrial vegetation compromises the health and viability of these ecosystems and their associated fauna (Groom *et al*. 2000).

The consequence of excessive recharge is familiar to most readers as the mechanism driving dryland salinity. Human disruption of the hydrological cycle via widespread land clearing, replacement of deep-rooted native vegetation with shallow-rooted crops and pasture, river regulation and irrigation all contribute to excessive recharge of aquifers. Rising water levels then intercept salt stored in previously unsaturated layers and transport it upwards, resulting in stream and land salinization. Halse *et al*. (2003) considered that the input of saline groundwater can pose a substantial threat to the biodiversity of surface wetlands and rivers and can drive shifts in faunal assemblage towards more salt-tolerant taxa. The thickness of the saturated zone and slow groundwater flow rates result in time lags before the impacts of pressures such as extraction or excessive recharge are apparent. Similar time lags can be expected before the efficacy of remediation efforts is known.

2.2.1 Climate change

In Victoria, climate change and climate change induced effects are not yet recognized as potentially threatening processes of direct relevance to freshwater biodiversity (Table 2). However, as noted in Section 2.1, climate is a fundamental landscape/catchment-scale driver of riverine structure, patterns and processes. It can therefore be expected to produce profound, cascading effects on riverine and terrestrial ecosystems. This section provides a brief account of the best available scientific data on current and expected future climatic conditions. I then consider what these projections might imply with respect to threat mechanisms/pathways.

Victoria has warmed by 0.6°C since the 1950s which is a faster rate of warming than the Australian average and the last ten years have been hotter than average in Victoria, with 2007 being the hottest year on record. Six out of Victoria's ten hottest years on record have occurred since 1990 (CES 2008).

Serious rainfall deficiencies over the past 11 years have reduced inflows to storages 30–60% below long-term averages. Water scarcity has been statewide in extent, exacerbated by high temperatures, and has worsened over time, with flow in the Murray and Melbourne storages reaching record lows in 2006 (CES 2008). The majority of Victoria is suffering serious deficiencies, and large areas (including the Port Phillip, Westernport, and Wimmera catchments) are experiencing the lowest rainfall levels on record. The majority of this rainfall decline is due to much drier autumns (Timbal & Murphy 2008). This is of concern because typically autumn rainfalls saturate the soil profile thereby creating the antecedent conditions that enables winter precipitation to generate runoff to rivers and streams. CES (2008) reported that Victoria has now experienced eight dry autumns in a row, and indeed 16 of the past 19 autumns have received below average rainfall.

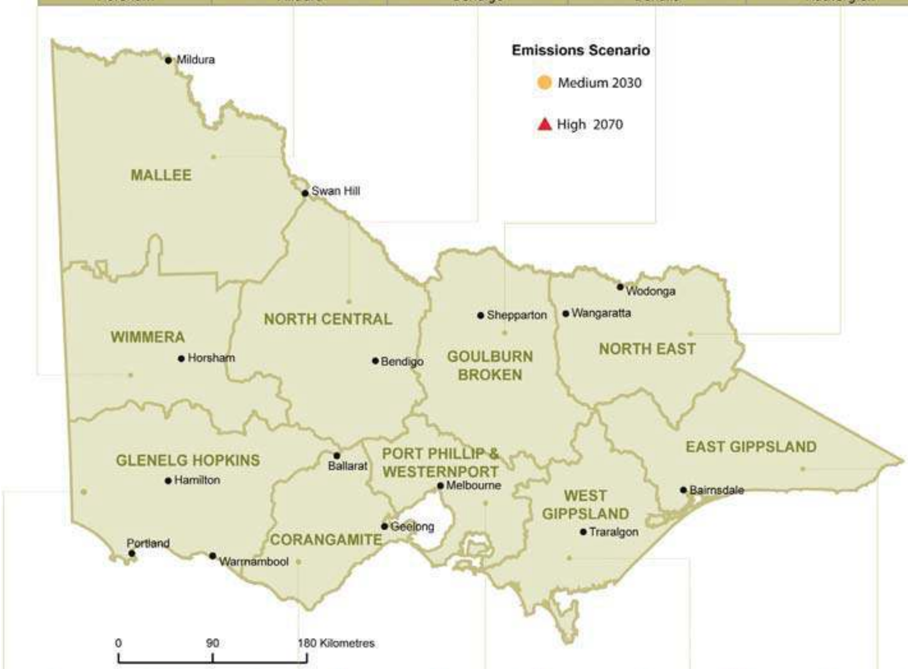
On behalf of the Victorian Government the CSIRO has prepared Statewide and regional climate change projections (DSE 2008). In this report, I focus on climate change projects for two time periods – 2030 and 2070. Essentially, the 2030 projections expected to be highly likely to occur as a function of previous and current level of greenhouse gas emissions. The 2070 projection is based on a high emissions growth scenario (known as the A1F1). This high emissions growth scenario assumes a continuation of strong economic growth based on continued dependence on fossil fuels and CO2 concentrations more than triple, relative to pre-industrial levels, by 2100. A global temperature increase of 4.0°C (2.4 to 6.4°C) is likely. This scenario represents the highest level of late 21st century emissions that were thought to be plausible back in 2000. However, recent evidence indicates that CO2 emissions have been growing at a more rapid rate (DSE 2008).

Modeling indicates that by 2030, the expected range of increase of average annual temperature in Victoria is +0.6°C to +1.2°C relative to the climate of 1990. This average incorporates average summer temperature increases of +0.6°C to +1.4°C and average winter increases of between +0.5°C and +1.0°C (CES 2008). The expected range of increase of average annual temperature by 2070 (under the high emissions scenario) is 1.8°C to 3.8°C (DSE 2008). The north and east of the State are expected to experience greater temperature increases than the south and west and seasonal analysis indicates that warming will be greatest in summer and least in winter (DSE 2008). Figure 2 summarizes climate change projections for Victoria's ten regions.

By 2030, annual average rainfall is expected to decrease by around 4% (range -9% to +1%). The greatest decreases in rainfall are likely to occur in winter and spring, while heavy rainfall intensity is most likely to increase in summer and autumn (DSE 2008). By 2070, average annual rainfall is expected to decrease by by 11% (range -25% to +3%) under the higher emissions growth scenario (DSE 2008). The greatest decrease in rainfall is likely to occur in spring. See Figure 2 for details for the various regions.

Since the early 1970s, Australian droughts have become more intense as a result of the warmer than average temperatures. The projections for warmer temperatures and reduced annual rainfall are likely to increase the risk of drought (DSE 2008). As well as decreases in total rainfall, evaporation is expected to increase, enhancing the overall drying trend. The annual average potential evaporation by 2030 is likely to increase by around 3% (1% to 5%), with the largest changes expected in winter. By 2070 evaporation could increase by 8% (range 2% to 16%) under the higher emissions growth scenario (DSE 2008). Refer to figure 2 for greater regional detail.

Climate Characteristic	Wimmera		Mallee		North Central		Goulburn-Broken		North East	
	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070
Average Temperature	+0.8°C	2.6°C	+0.9°C	2.8°C	+0.9°C	2.8°C	+0.9°C	2.8°C	+0.9°C	2.9°C
Average Rainfall %	-4%	-12%	-4%	-11%	-4%	-11%	-3%	-10%	-3%	-10%
Potential Evaporation %	2%	8%	2%	7%	2%	8%	3%	8%	3%	9%
Relative Humidity %	-0.70%	-2.10%	-0.70%	-2.20%	-0.70%	-2.30%	-0.70%	-2.30%	-0.60%	-2.00%
Wind Speed %	no change	no change	no change	no change	no change	-1%	-1%	-2%	-1%	-2%
Frost Days	23	9	14	5	26	11	35	17	65	40
	Horsham		Mildura		Bendigo		Benalla		Rutherglen	



Climate Characteristic	Glenelg-Hopkins		Corangamite		Port Phillip-Westernport		West Gippsland		East Gippsland	
	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070	Medium 2030	High 2070
Average Temperature	+0.8°C	2.4°C	+0.8°C	+ 2.4°C	+0.8°C	+ 2.6°C	+0.8°C	2.6°C	+0.8°C	2.7°C
Average Rainfall %	-4%	-13%	-4%	-12%	-4%	-11%	-4%	-11%	-3%	-9%
Potential Evaporation %	3%	8%	2%	8%	3%	9%	3%	10%	3%	10%
Relative Humidity %	-0.50%	-1.70%	-0.50%	-1.70%	-0.60%	-2%	-0.50%	-1.70%	-0.50%	-1.50%
Wind Speed %	no change	no change	no change	no change	no change	no change	no change	no change	no change	no change
Frost Days	16	5	33	14	2	0	25	10	11	2
	Hamilton		Ballarat		Melbourne		Sale		Orbost	

Figure 2. Summary of climate change projections for 2030 and 2070 for each of Victoria's ten regions. (Source: 'State of the Environment – Atmosphere' (CES 2008, p.209).

Projected changes in rainfall and higher rates of evaporation will result in less runoff and less water for our catchments, dams and rivers. By 2030, catchments located in the north east and south east may experience up to 30% reductions in runoff, those in the north west can expect decreases ranging from 5% to 45% while the southwest can expect 5% to 40%. By 2070, runoff into catchments in East Gippsland may increase by 20% or decrease by 50% depending on changes in rainfall. The rest of the state can expect declines of at least 5% or up to 50%.

By 2030, streamflow may vary from no change or slight increases in East Gippsland to 25-40% decreases in river systems in western and north-western Victoria. By 2070, streamflow may decrease by up to 50% across much of the State (CES 2008-inland waters). With lower water availability across most of Victoria, soil moisture levels, streamflow and recharge of groundwater are all likely to decrease. As Victoria's growing population is heavily dependent on surface water (and increasingly groundwater) sources, reduced water availability is likely to intensify competition for water resources and exacerbate alteration of natural flow regimes.

The warmer, drier weather is also likely to increase the frequency and intensity of bushfires. Fire risk is forecast to increase substantially in Victoria with the number of very high or extreme fire danger days across southeastern Australia expected to increase by up to 25% by 2020 and up to 230% by 2050 (CES 2008). This has implications for water quality and the health of instream biota. There is also concern that (large-scale) post-fire regrowth of vegetation will alter catchment hydrology as growing forests utilize more water than mature forests and could further reduce runoff (CES 2008).

Higher air temperatures can increase water temperatures which produces a range of physical and physiological effects on instream biota (Caissie 2006). Temperature affects fundamental ecological variables, such as oxygen concentration, respiration, production and decomposition. High water temperature reduces the capacity of water to carry dissolved oxygen while also increasing the rates of respiration of aquatic organisms and decomposition. The heating and expansion of surface layers also increases the risk of thermal stratification in stationary water bodies (e.g. dams, weir pools, billabongs) or very slow flowing river reaches. This combination of effects is already common over summer low flow periods in many Victorian rivers and streams (e.g. lower Wimmera River and Broken Creek) and is of concern as it can lead to a host of problems including hypoxia/anoxia of bottom waters, aquatic habitat degradation, increased concentrations of heavy metals and toxicants such as ammonia and hydrogen sulphide in deoxygenated waters and increased risk of algal blooms (Chee 2005). This is particularly deleterious for freshwater biota as these environments may represent a major habitat resource over the summer drought or low flow period.

Collectively, the forecast for higher temperatures, reduced rainfall, increased evaporation, reduced soil moisture levels, runoff, streamflow and groundwater recharge all affect components of the hydrological cycle and ultimately the natural flow regimes and ecological functioning of surface water and groundwater ecosystems. However, the fact is that past river regulation and land management practices have already wrought profound changes on natural flow regimes and continue to do so. Indeed, McMahon and Finlayson (2003) consider that "the changes brought about by the regulation of rivers are much more rapid and dramatic than those which might occur as a result of climate change". Furthermore, they suggest that even if the most extreme predicted climate change scenarios for Australia were to eventuate, their impact and rate of onset, at least for surface waters, would be on a lesser scale than the changes that have already occurred as a result of river regulation. With respect to groundwater systems, Crosbie (2007) suggested that the direct impact of climate change on groundwater will be small relative to the impacts induced by pressures associated with development such as urbanization and increasing consumptive demand. These views emphasize the severity of existing threatening processes and it is likely that they will be further exacerbated by climate change induced effects.

3 Freshwater Spatial Conservation Prioritization

The object of this section is to review key concepts and techniques in reserve system design and spatial conservation prioritization. First, I provide an exposition of the Comprehensive, Adequate and Representative or CAR principles for reserve system design and an analysis of what they mean with respect to freshwater conservation in Victoria. I then introduce the modern framework for spatial conservation prioritization and explain the suite of interrelated techniques involved.

3.1. CAR Principles for Reserve System Design

The principles of **Comprehensiveness**, **Adequacy** and **Representativeness** were originally crafted with regard to Australian forest ecosystems. As the CAR principles have come to be identified as the foundation for reserve design in Australia, a clear understanding of the statement, intent and implications of these principles is important. Box 1 presents the CAR principles as originally described in Commonwealth of Australia's (1997) 'Nationally Agreed Criteria for the Establishment of Comprehensive, Adequate and Representative Reserve System for Forests of Australia'.

Box 1. Principles of comprehensiveness, adequacy and representativeness for the establishment of a reserve system for forests of Australia (excerpts from Commonwealth of Australia 1997).

Comprehensiveness - includes the full range of forest communities recognised by an agreed national scientific classification at appropriate hierarchical levels (NFPS 1992).

This principle requires that the reserve system samples the full range of forest communities across the landscape. However, the wide variation in forest ecosystems across the continent, and the large gaps between forested regions, makes effective consideration of comprehensiveness on a continental scale most difficult. Smaller and more manageable regional units are therefore necessary as a basis for consideration of comprehensiveness. Forest ecosystems, forest types and forest vegetation communities, together with their environmental descriptors, are commonly used as surrogates for biodiversity and as a basis for planning a comprehensive reserve system. [It is acknowledged that each of the terms may have different meanings across different jurisdictions in Australia and hence,] surrogates used to assist with establishing the CAR reserve system will need to be determined on a regional basis.

Adequacy - the maintenance of ecological viability and integrity of populations, species and communities (NFPS 1992).

Adequacy addresses the difficult question of extent: what is the level of reservation that will ensure viability and integrity of populations, species and communities. There are many approaches, ranging from best-guess estimates for poorly-defined ecosystems, to very accurate calculations for endangered or specific populations of animals and plants. Where data on the viability of populations are available, they should be incorporated in determining the adequacy of a reserve system.

No precise basis exists for determining criteria that provide for adequacy. However, the general rule is that the chances of long-term survival increase with increased proportions of populations or forest ecosystems reserved and appropriately managed. The degree of risk varies with different species (or suites of species) and with the degree of modification of the contiguous native forest beyond reserves. Most estimates show that the risk of loss is highest where only a small percentage of the distribution of the community or species is reserved and adjoining unreserved forest is cleared or significantly modified.

Replication across the range of geographic, environmental and biotic domains must also be considered when determining the adequacy of the reserve system. Replication is essentially "insurance" against the loss of natural values due to stochastic events (such as fire) which may dramatically reset successional processes and reduce or entirely remove key habitats. Implicit in the maintenance of biodiversity is the requirement to sustain ecological processes and functions and provide for the maintenance of natural patterns of speciation and extinction. This requires that the adequacy of a reserve system be considered in a landscape context (e.g., Saunders and Hobbs, 1991). The extent of inclusion of whole catchments, the degree of sympathetic management of adjacent lands, and the options for provision of corridors to provide linkages are important in the development of integrated nature conservation strategies. Factors operating within the surrounding landscape that are particularly relevant to determining the adequacy of the reserve system are threatening processes (e.g., land clearing and disease), and the conservation strategies adopted in forests outside those areas reserved specifically for conservation.

Representativeness - those sample areas of the forest that are selected for inclusion in reserves should reasonably reflect the biotic diversity of the communities (NFPS 1992).

This principle is designed to ensure that the diversity within each forest ecosystem is sampled within the reserve system. Many species, particularly animals, have distributions that are not easily predicted by surrogates such as forest ecosystems, and information on species distributions and genetic variation should be used in reserve design. There are good distributional data for a large number of forest species, genotypes and communities, and reserve selection methods such as described in Kirkpatrick (1984), Margules et al (1988) and Pressey and Nicholls (1989) can be used to ensure that all species whose distributions are relatively well known are represented in the reserve system. The focus of these methods should be on those species that that depend on reservation for protection.

Using species distributions alone will not guarantee the inclusion of all elements of biodiversity. However, using these distributions together with other measures of forest diversity can increase confidence that the reserve system does cover the full range of biodiversity. Other measures of forest diversity may include, for example, the occurrence of a vegetation type in relation to different soil types or the variation in structure and floristics present within a forest ecosystem. In practice a combination of approaches needs to be used to assess the representativeness of the reserve system.

It is not necessary to ensure that every element of biodiversity that occurs within a forest ecosystem is reserved within that ecosystem. Many species may be well represented in one forest ecosystem in a region and infrequent in another, and it is not necessary to distort reserve boundaries to ensure that they are reserved in each ecosystem occurrence. The important thing is that known species and genotypes are adequately reserved with the aim of maximising their viability within a region, not that they are reserved in every forest ecosystem in which they have been recorded.

Representativeness should be approached in a practical way. Available or readily acquirable data, depending on its type, quality, and resolution, should be used in the design of a reserve system

How might these principles be applied to the protection and conservation prioritization of freshwater biodiversity in Victoria? Firstly, application of the ‘comprehensiveness’ principle pre-supposes the existence of an exhaustive, well-defined and detailed inventory of the communities of interest, preferably including spatial definition so that they can be mapped. In contrast to the situation with terrestrial vegetation, freshwater biotic communities have not been identified, classified and described in a systematic and consistent manner across the State. Victoria has broad classifications of types of freshwater environments but these are constructed wholly from biophysical attributes (i.e. the classification schemes do not explicitly include or account for biological data).

There appears to be good quality, fine-scale (roughly 1:25,000) data for wetlands. Corrick and Norman (1980) developed what has become the most widely-used wetland classification system in Victoria. The system has nine categories based on water depth, water permanency and salinity. Victoria’s wetlands have been mapped and classified using the Corrick and Norman (1980) system and two spatial data (GIS) layers have been developed by DSE for pre-European settlement and wetlands mapped from 1975-1994 (datasets named, WETLAND_1788 and WETLAND_1994 respectively, DSE Corporate Geospatial Data Library).

With respect to lotic riverine systems, the ‘Rivers & Streams Special Investigation’ (LCC 1991), collated the best available data from recent surveys, published literature, unpublished reports, submissions from government departments, public authorities, and interested individuals and organizations. They also developed and applied a simple river classification system based on a combination of geomorphologic units (as defined by Jenkin & Rowan 1987) and hydrological regions (as defined by Hughes & James 1989). With this classification scheme, LCC (1991) identified 16 different river-catchment types and selected representative rivers that typified its river-catchment type.

The Rivers & Streams Special Investigation only accounted for rivers and streams that were (Strahler order) third-order or greater, with stream order calculated from the drainage network of the 1:250 000 map series for Victoria (LCC 1991). In spatial terms, 1:250,000 is fairly coarse-scale and coupled with the threshold of third-order (or greater) means that small streams and tributaries were omitted from the assessment. The Jenkin & Rowan (1987) geomorphic units used for classification numbered only nine and the Hughes & James (1989) hydrological regions numbered only five (as hydrological characterization was constrained by the availability of flow gauges with records of adequate length and quality). As Janet Stein (*pers. comm.*, cited in Nevill & Phillips 2004) pointed out small streams and minor tributaries constitute by far, the majority of total stream length. They often constitute qualitatively different types of riverine environments relative to major rivers and streams and have a major influence on the characteristics, ecological functions and processes of receiving waters. For all these reasons, it is questionable if the Rivers and Streams Special Investigation was sufficiently comprehensive and a more detailed update is probably warranted.

The principle of adequacy seeks to ensure to maintenance of ecological viability and integrity of populations, species and communities. Adequacy however, is a means objective in service of a more fundamental objective, which is the long-term persistence of these species, populations and communities. Commonwealth of Australia (1997) points out that "No precise basis exists for determining criteria that provide for adequacy". This reflects the fact that a scientifically rigorous approach to determining adequacy for the purpose of ensuring persistence is a highly complex, multifaceted task that requires grappling with:

a) the specific characteristics of the biological entity (i.e. species, population, community) such as distribution, abundance, habitat and life-history requirements, reproductive capacity, rates of survival and establishment and dispersal abilities;
b) the identification of particular ecological functions and processes that are necessary to sustain/fulfil any aspect of (a);
c) dynamic threats/threatening processes acting either directly or indirectly (such as in the upstream-downstream watersheds or in surrounding regions adjacent to channels and floodplains) on any aspect of (a) and (b); and perhaps
d) conservation actions taken to ameliorate (c)

The challenges of ensuring persistence in the face of threats is considered in greater detail in § 3.2.3.1.

Finally, the principle of 'representativeness' essentially acknowledges that the 'forest ecosystem types' or communities identified at broad, regional scales as a basis for consideration of comprehensiveness contains finer-scale biotic heterogeneity that warrants specific attention to ensure that it is explicitly accounted for in reserve design. The principle of 'representativeness' as described in Commonwealth of Australia (1997) recognizes that coarse-scale surrogates such as 'forest ecosystem' types/communities are useful, but not necessarily sufficient when it comes to predicting the occurrence of lower-order entities in the hierarchy of ecological organization (e.g. vegetation communities of distinctive floristic composition, species or populations of species with different genotypes/phenotypes etc). They advocate the use of finer-scale data such as distributional data on "species, genotypes and communities" particularly in cases where the biological entity depends on reservation for protection. But they do not provide firm guidelines or prescriptions for implementing 'representativeness' and instead emphasize practicality.

Recently however, quantitative spatial tools have been developed for the state of Victoria and Victoria now has detailed, fine-scale, spatially explicit environmental and biological distributional data for the freshwater realm that can be profitably employed to address both 'comprehensiveness' and 'representativeness'. These are described in § 3.2.2.3.

3.2 Spatial Conservation Prioritization

In this report, the term 'spatial conservation prioritization' refers to the application of quantitative techniques to generate spatially explicit information about conservation priorities (*sensu* Moilanen *et al*. 2009d). A key output of the process is the identification of a set of priority areas for conservation investment or action. Conservation action or investment can take many forms such as protected area designation, acquisition for conservation, amelioration of threatening processes and restoration.

Much of the material in this and following sections is based on the seminal paper on 'Systematic conservation planning' by Margules & Pressey (2000) and a recent edited volume on 'Spatial Conservation Prioritization' (Moilanen *et al*. 2009a) which provides state-of-the-art accounts on all facets of the conservation prioritization problem.

Spatial conservation prioritization proceeds via a series of interrelated questions that a conservation planner poses and the (sometimes iterative/recursive) tasks that arise from the set of questions (Table 3). These questions define the essential components of any spatial conservation prioritization problem which are as follows (after Margules & Pressey 2000; Wilson *et al*. 2007; Moilanen *et al*. 2009b):

1. Clearly define a conservation objective and use this to define an explicit measure of performance (e.g. maximize the number of species conserved).
2. Characterize the spatial distribution of biodiversity features or surrogates for these features (e.g. patterns of species distribution or habitat types or landscape classes across the landscape)
3. Identify and quantify other conservation-relevant considerations (e.g. spatial arrangement of planning units, connectivity and vulnerability to threatening processes)
4. Define the resource constraints (e.g. total available budget)
5. Specify the set of conservation actions that can be taken, as well as the costs associated with each (e.g. reserve establishment, control of invasive species, replanting)
6. Specify how actions contribute to the defined conservation objective (e.g. how many species will be conserved if an area is protected or if an area is restored)

Prioritization (or optimization) is then a process of selecting actions for areas, so as to achieve a solution that gives the highest possible value (according to the performance measure), whilst satisfying all constraints.

These questions and tasks in Table 3 provide the organizational structure for report sections that elaborate on what each of the major tasks involves and particular considerations and challenges that ought to be taken into account. Figure 3 gives an overview of the flow of analysis and the linkages between problem components/major tasks.

Table 3. Guiding questions for spatial conservation prioritization and corresponding tasks. Final column indicates the report section that discusses considerations associated with each major task.

	Question	Task	Section
A	What are the biodiversity features we care about? (e.g. genes, species, communities, habitat types) What should our goals be and how do we measure progress in reaching them?	Goals & Objectives	§ 3.2.1
B	What is the distribution of these biodiversity features in the focal landscape/region?	Identifying Candidate Planning Units: Data	§ 3.2.2

	Which are the areas of particular importance for our biodiversity features?	Collection, Measurement, Modeling & Mapping	
C	Can we identify and quantify characteristics of candidate planning units that are salient for conservation? (e.g. landscape context, spatial arrangement and connectivity, size, shape, condition and vulnerability to threatening processes.)	Characterizing Candidate Planning Units	§ 3.2.3
D	What resources do we have at our disposal? Or conversely, what constraints do we need to operate within? What is the set of conservation actions under consideration? (e.g. protected area designation, amelioration of threatening processes, restoration or some combination.) How do conservation actions affect the outputs of question C and (in turn) B?	Constraints & Conservation Actions	§ 3.2.4
E	How do we select a set of priority areas using the results for questions A, B, C & D (e.g. scoring vs. complementarity)	Selection Strategies & Optimization	§ 3.2.5

Our assumptions, simplifications, choices and decisions for each task can produce many variants of the conservation prioritization problem. The more closely they resemble real-world situations, the more likely they are to be conceptually and computationally complex. For example, multiple versus single features and objectives, incorporation of habitat quality, connectivity and multiple threatening processes and multiple versus single conservation actions and their interactive effects on (multiple) features/objectives. By the stage represented by question E (Table 3), the problem is likely to have attained dimensions that exceed unaided cognitive abilities. In such situations, we need to turn to quantitative methods to cope and render such problems tractable. To this end, spatial conservation prioritization has borrowed heavily from the field of classical optimization which has provided a useful framework and solution techniques (see Moilanen *et al*. 2009b for further details).

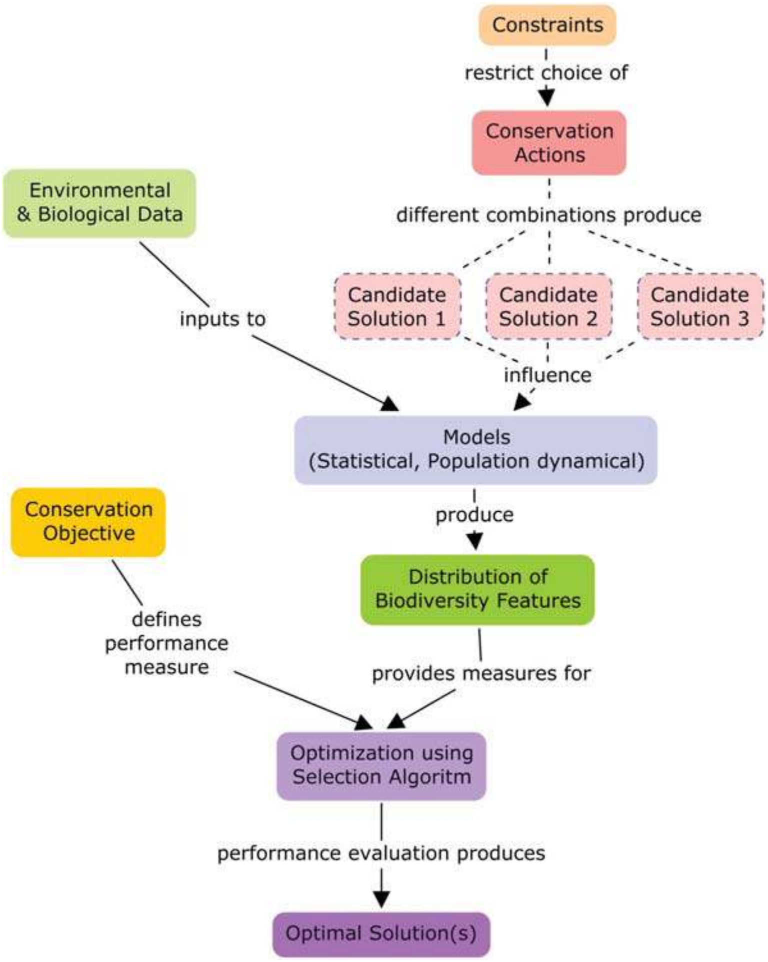


Figure 3. Schematic showing flow of analysis and how components of the general prioritization task link together (after Moilanen *et al.* 2009b).

3.2.1 Goals & Objectives

Precise specification of conservation goals is a critical step in formulating the conservation 'problem'. In order to implement analytically rigorous spatial conservation prioritization, conservation goals must be measurable, in other words, they must be capable of being expressed in quantitative terms (which we call the 'objective'). Ferrier & Wintle (2009) explain why this can be a challenging task:

a) there is no widely accepted typology or terminology for conceptualizing and formulating such goals because they derive from societal judgements of value.
b) Such values can range widely from the relatively tangible (e.g. market value and option value) to intangible (e.g. intrinsic, non-use value and existence value – see Chee 2004 for further details).
c) Translating any such values into conservation/planning goals is complicated by the fact that the notion of biodiversity itself is multi-faceted, encompassing multiple levels (genes, species, communities, ecosystems) and dimensions (composition, structure, function) of organization.

The simplest goals to specify generally relate to individual biological entities of particular conservation concern, such as high-profile threatened or endangered species. Another common goal focuses on species-level compositional diversity and the assumed objective is to maximize the number of species retained in the region of interest. One must however, be cognizant of the fact that when the focus of assessment/analysis is shifted across levels or dimensions of biodiversity, for example, to account for genetic, community or functional diversity, the resultant priorities could be markedly different from that based on raw numbers of species (Ferrier & Wintle 2009).

Managers (possibly dominated by concerns about institutional obligations they are mandated to achieve and institutional constraints they are obliged to operate within) often tend to frame goals in broad, aggregate and inclusive terms such as all biodiversity - meaning all native, endemic species within the region of interest (see e.g. Chee 2005). There is also growing interest in defining conservation goals in terms of ecosystem services (Chan *et al*. 2006; Egoh *et al*. 2007). Some of the challenges associated with this include the following:

a) like biodiversity, ecosystem services is a multi-faceted concept. Ecosystem services arise from biophysical and ecological processes (see § 2.1.1) that are highly interconnected, interactive in nonlinear ways and subject to stochastic influences at a range of spatiotemporal scales. There are often interdependencies between various types of ecosystem services which means that it may be impossible to parse certain services into independent conditions and processes for use in assessment (Chee 2004).
b) assuming that relatively independent ecosystem services can be identified, how should one select which to give emphasis to (e.g. hydrological regulation vs provision of water for irrigation and livestock vs biogeochemical cycling vs recreation opportunities vs provision of aesthetic beauty, cultural, intellectual and spiritual inspiration) given that the choice(s) will influence resultant priorities.
c) data and methods enabling ecosystem services to be measured and mapped is frequently lacking (Egoh *et al*. 2007) which is not surprising given the complexity of some types of ecosystem services, for instance, biogeochemical cycling.
d) how well would priorities identified on the basis of ecosystem services concord with those based on biodiversity measures? As Naidoo *et al*. (2008) show, concordance may turn out to be poor (i.e. ecosystem service-based priorities might be poor indicators of biodiversity-based priorities and vice versa) because areas important for ecosystem services might not necessarily be important for biodiversity. This might occur because many types of ecosystem services are largely provided or regulated by relatively common and abundant biological entities, whereas biodiversity conservation is frequently concerned with rarer or threatened/endangered entities (Chan *et al*. 2006). It is also worth noting that many types of ecosystem services that society values highly, for instance, erosion control and river bank stabilization and recreational opportunities for fishing and boating can be fulfilled by alien species and management practices which are inimical to biodiversity conservation. For example, erosion control and river bank stabilization can be provided by alien Willow species and practices such as 'rocking' river banks which interferes with natural geomorphic processes that maintain channel complexity and physical habitat structure for riverine biota. Recreational fishing opportunities can be provided by alien species such as brown trout *Salmo trutta*, rainbow trout *Onchorynchus mykiss* and redfin *Perca fluviatilis*. And boating opportunities are enhanced by practices such as desnagging which detracts from both the provision and maintenance of habitat processes for freshwater biodiversity.

Philips & Butcher (2005) championed an ecosystem services-based approach to riverine conservation planning in their report on 'River Parks: Building A System of 'Habitat Management Areas' Across the Murray-Darling Basin'. They argued that an ecosystem service perspective which extends beyond biodiversity conservation to embrace river health, social and recreational benefits is more likely to attract community support and engagement. However, they do not explain which ecosystem services ought to be focused on, nor how to measure or map them in order to identify priority 'Habitat Management Areas'. While I acknowledge the benefits of an ecosystem service perspective, there are clearly many issues that need to first be addressed.

The main point from the foregoing discussion is that the choice of conservation goals and the specification of the relative importance of the goals selected is non-trivial and has significant ramifications for the methods used in, and the results obtained from, spatial conservation prioritization. This foundational step in any assessment of conservation priorities would clearly benefit from consultation with all stakeholders. At the very least however, they should be clearly and explicitly defined and documented to ensure transparency of the subsequent assessment as well as establishing the basis for accountability.

3.2.2 Identifying Candidate Planning Units: Data Collection, Modeling & Mapping

In this stage, the main tasks are to: a) delineate appropriate planning units; b) gather the raw ingredients required to characterize pattern (and perhaps process where possible); c) produce spatially explicit maps of distribution of target biodiversity features; and d) collect any other relevant spatial data that can inform the subsequent stages of the assessment process (e.g. existing reserves, road networks, land tenure and land use potential).

3.2.2.1 Delineating planning units

Planning units are the basic spatial units for mapping biodiversity patterns, characterizing and quantifying factors such as condition and vulnerability to threatening processes and evaluating conservation actions on biodiversity. In other words, planning units are the fundamental spatial entities on which the assessment operates. In terrestrial studies, planning units are generally regular (e.g. grids or hexagons) although irregular planning units such as polygons and watersheds are also used (Margules & Pressey 2000). It is generally recognized that grid/hexagon-based planning units are inappropriate for river networks which are influenced by topographically-defined catchment areas and are spatially connected in both longitudinal and lateral directions. The small number of published studies of freshwater spatial conservation planning/prioritization (e.g. Linke *et al*. 2007, 2008; Thieme *et al*. 2007; Moilanen *et al*. 2008; Amis *et al*. 2009; Nel *et al*. 2009) use subcatchments (delineated using automated hydrological processing routines in GIS software) as the planning unit to ensure that spatial relationships/dependencies are captured.

With the exception of Moilanen *et al*.'s (2008) study however, planning unit sizes for the studies mentioned above ranged from 10s to 100s of km^2. It is important to consider if the spatial resolution of defined planning units is appropriate for and compatible with the purpose of the study and all the tasks in the conservation prioritization assessment process. Large areas invariably encompass a degree of heterogeneity. Does the scale make sense for mapping biodiversity patterns of target biodiversity features (e.g. benthic invertebrate, amphibian and fish species)? How does one quantify the condition of a subcatchment extending 100s of km^2 which may include multiple land use types and spatial variation in water resource development and management? Planning unit sizes ranging from 10s to 100s of km^2 might suffice for provisional and/or broad-scale conservation prioritization (e.g. Thieme *et al*. 2007; Klein *et al*. 2009), but is arguably too coarse relative to the scales at which on-ground managers are generally obliged to operate (e.g. Cowling *et al*. 1999; see also Ferrier & Wintle 2009). For instance, the conservation action of restoration is rarely carried out at the scale of 10s to 100s of km^2.

3.2.2.2 Modeling & mapping biodiversity patterns

Approaches to (modeling and) mapping biodiversity pattern (and process where feasible) form a continuum with wholly environmentally-based methods at one end and biologically-based methods on the other end. Some degree of expert input/interpretation is common to all approaches. The first step in either approach (or hybrid approach) is the collation of relevant data. The first requirement is that the data must be spatially explicit, in other words, referenced to geographic locations ('georeferenced') and therefore capable of being mapped. Spatial environmental data is commonly sourced from various forms of remote sensing (e.g. satellite data, radar data, airborne scanners, aerial photography) and digitized extant analog mapped products (e.g. topographic, vegetation, geology and soil maps). Relevant spatial data is usually prepared and manipulated within a GIS and spatial interpolation and modeling used to fill gaps in spatial coverage. Once again, the spatial resolution of source data needs to be appropriate for and compatible with the intended use and data preprocessing and quality-control measures are also important to ensure data accuracy, consistency and overall quality. Together, these will determine the degree of confidence users can place on the outputs.

Predominantly environmentally-based methods are often used when there is a perceived lack of availability of reliable georeferenced biological data (for freshwater examples, see e.g. Higgins *et al*. 2005; Thieme *et al*. 2007). The underlying premise is that habitats, ecological processes and biodiversity are shaped by abiotic attributes such as bioclimatic, physiographic, edaphic and hydrologic variables, acting at hierarchical spatiotemporal scales. So classes derived from some form of pattern/numerical analysis of abiotic environmental attributes may serve as surrogates for spatial patterns of biodiversity. As Ferrier *et al*. (2009) note, the limited testing of this assumption to date (e.g. Ferrier & Watson 1997) "has yielded mixed results, suggesting that the level of concordance between environmental and biological patterns probably depends on the type of environment involved, the geographical extent and resolution of assessment, and the biological group of interest.". Environmental classification is also variously known as 'environmental domain analysis', 'regionalization', 'bioregionalization', 'ecoregionalization', 'biogeographic regionalization' and most recently, for Australian freshwater systems, 'ecohydrological regionalization' (Pusey *et al*. 2009).

Ideally, the environmental attributes used in classification should have functional relevance for biodiversity features of interest. For instance, soil type and seasonal measures of temperature, precipitation and solar radiation for terrestrial vegetation; measures of temperature, light availability and currents at different depths for marine biota; and seasonal measures of temperature, hydrological indices, channel geometry and substrate types for freshwater biota. Another important consideration is that the temporal stability of the attribute will necessarily affect the temporal stability of the resultant classification. Therefore, chosen attributes should be relatively stable over time, or should already incorporate temporal variation. Alternatively, the classification could be updated over time (Kingsford *et al*. 2005).

An example of a wholly environmentally-based classification is the hydrogeomorphic classification used in the Rivers & Streams Special Investigation (LCC 1991) mentioned in § 3.1. The Interim Biogeographical Regionalisation of Australia (IBRA) is an example of a predominantly environmentally-based classification which includes some biological data. The most current version is IBRA 6.1, developed in 2004. IBRA 6.1 includes attributes of climate, geomorphology, landform, lithology and characteristic flora and fauna (unfortunately, no easily accessible details on what these are). This broad, continental-scale landscape classification produced 403 sub-regions to assist with National and State approaches for developing a consistent system of CAR terrestrial reserves. 21 IBRA sub-regions fall wholly or partially within the Victorian state boundary. A recent update appears to have been conducted by DSE in Victoria, with a total now of 28 bioregions defined and mapped (Figure 4).

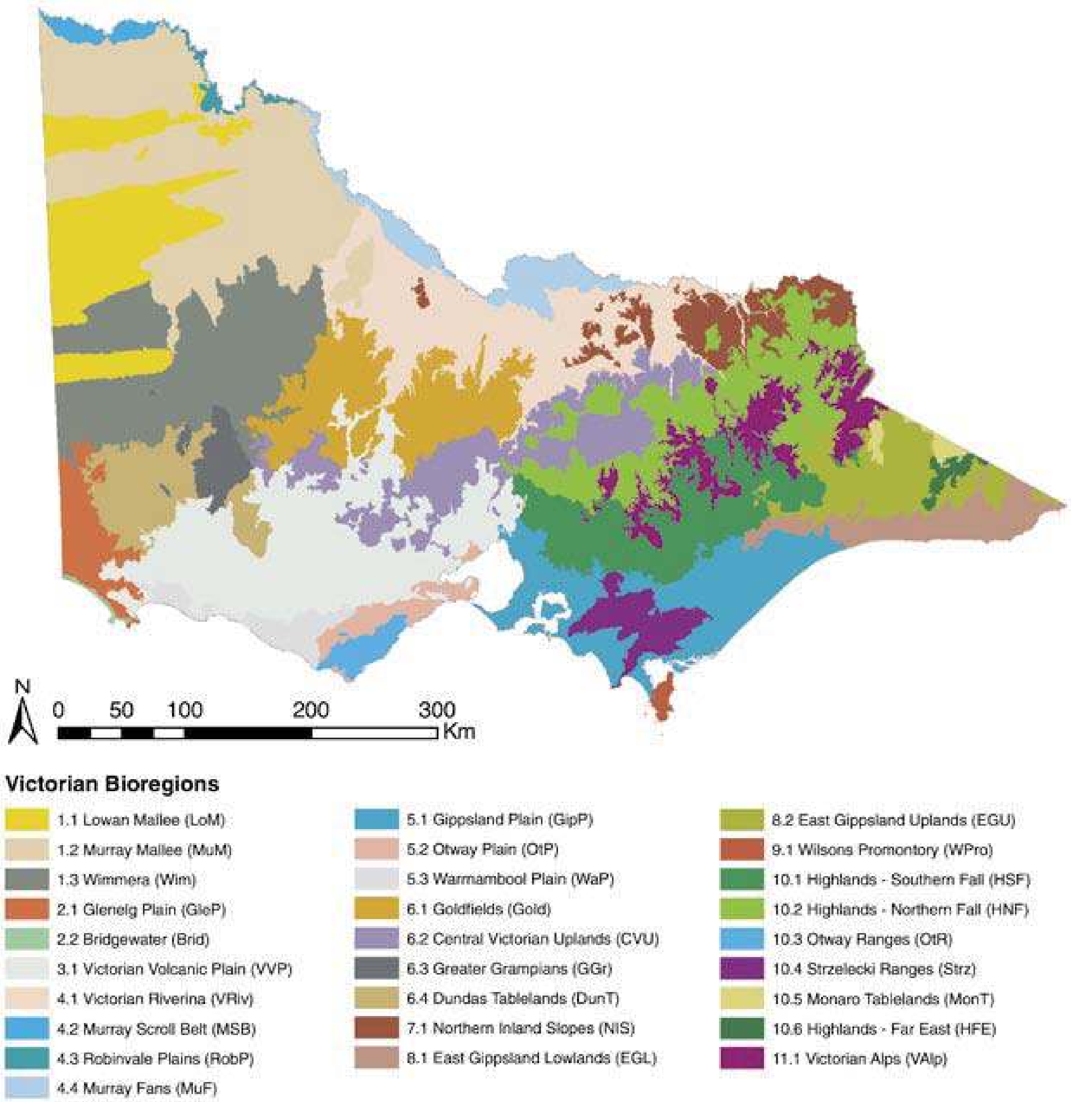


Figure 4. Victoria's 28 bioregions.
(Source: www.dpi.vic.gov.au/DPI/Vro/map_documents.nsf/pages/bioregional_strategic_overviews)

The principal disadvantages of an IBRA-style classification for freshwater ecosystems are as follows:

a) a terrestrially-focused classification scheme generally does not include adequate environmental data of functional relevance to riverine ecosystems (e.g. hydrological data) and this may affect the level of concordance between environmental and biological patterns.

b) too coarse a spatial resolution - environmental data is aggregated/averaged within spatial units (IBRA sub-regions/Victorian bioregions) that are very large (100s to 1000s of km^2), relative to the spatial resolution required to usefully categorize spatial variation of riverine habitats such as rivers, streams, lakes, wetlands and aquifers (Nevill & Phillips 2004), to say nothing of capturing spatial patterns of riverine biodiversity which are influenced by abiotic variation at much finer spatial scales. There is also a serious scale mismatch between such broad-scale classes and the scale at which on-ground land use decisions and management actions are typically contemplated and implemented.

More promising approaches for riverine ecosystems include New Zealand's River Environment Classification (REC, Snelder *et al*. 2004a, 2005) and Pusey *et al*.'s (2009) 'Ecohydrological regionalization of Australia'. Irrespective of the approach used, it is important to establish if the classification is any good. In other words, it is critical to evaluate and validate the performance (accuracy) and utility of all classifications with specific reference to the intended purpose(s). Snelder *et al*. (2004b) provide an example of such an evaluation with respect to the REC's ability to explain spatial variation in aquatic invertebrate assemblages. Snelder *et al*. (2005) provide an example of the REC's ability to explain variation in the hydrological character of rivers.

As we are specifically concerned with biodiversity however, an approach to mapping biodiversity patterns that is informed by actual biological data is likely to be more credible and satisfactory. When available, georeferenced biological (usually species presence-absence, presence-only or abundance) data is usually in the form of point location data. Such data might be obtained from surveys, records in museum collections and government and research institutions. When sourcing biological data for use in mapping contemporary biodiversity patterns, it is important to be mindful of data currency. If there have been substantial changes to the natural environment, the use of historical data for mapping biodiversity pattern may result in serious inaccuracies that can undermine the conservation prioritization assessment. Point locality data can be used directly for mapping and assessment (see Roux *et al*. 2008, Amis *et al*. 2009 and Nel *et al*. 2009 for examples in freshwater environments). However, such data tends to be geographically sparse and moreover, biased towards more easily accessible areas. With present technologies available for spatial modeling, we can do better than that.

Species distribution modeling (SDM) provides a valuable and powerful way of filling gaps in spatial coverage. SDMs are quantitative tools developed from observations of species occurrence or abundance and spatial environmental data. Essentially, the process involves analyzing relationships between species and environment attributes and formalizing them in a statistical model. As intimated in the discussion on environmental classification, the environmental predictors (attributes) used should be ecologically-meaningful/functionally relevant for the target species (Elith & Leathwick 2009a,b). These models can then be used to predict the distribution of the focal species in unsampled regions (assuming the requisite spatial environmental data exists for those regions). By explicitly linking biological data with environmental characteristics, SDMs represent a more rigorous, evidence-based approach to mapping biological pattern. SDM is a well-developed research area and SDMs are now widely used across terrestrial, freshwater and marine realms. For accessible, comprehensive and detailed expositions on key considerations, methods and steps involved in SDM, the reader is referred to Elith & Leathwick (2009a, b).

3.2.2.3 Modeling & mapping freshwater biodiversity patterns in Victoria

The development of SDMs for freshwater biota can be challenging because their occurrence (and abundance) patterns typically exhibit complex, non-linear relationships to physical habitat heterogeneity and biotic interactions. Riverine habitats are inherently heterogeneous and nuanced, shaped as they are, by the interactions of flow magnitude, frequency, timing and duration with the geomorphic template. Additional complexity is overlaid by the effect of upstream and downstream features along a flow path (such as dams, weirs or waterfalls) and influences arising from bioclimatic processes and anthropogenic activities in the upstream contributing catchment area (see § 2.1). Nevertheless, the requisite biological and environmental data exists and spatial modeling 'infrastructure' for river networks across the entire State has recently been developed.

Spatial data for freshwater taxa in Victoria is available from a range of sources including DSE's Aquatic Fauna Database (AFD), the Environment Protection Authority (EPA), the Arthur Rylah Institute for Environmental Research, the Sustainable Rivers Audit program for the Murray-

Darling Basin, scientists in university research institutions and various sources in published and 'grey' literature.

A fine-scale, comprehensive stream network GIS database has recently been completed for the state of Victoria (Chee, unpublished, see Appendix 1 for details). The backbone of the database, is an ordered, link-node representation of the stream network for Victoria. The stream link (i.e. the river segment between any two stream junctions) constitutes the basic 'unit' in the network. In order to enable characterization of the different aspects of the complex environmental space arising from longitudinal and lateral mosaic influences on riverine ecosystems (see § 2.1), a comprehensive set of estimates of physiographic, bioclimatic, edaphic, land cover and anthropogenic disturbance-related variables, which were considered to have ecological relevance, were computed for every stream link at one or more, hierarchically-nested spatial scales (Appendix 1, Table 1). The three scales were: a) the riparian zone with a width of 50 m on either side of a link; b) the immediate watershed of a link and c) the entire upstream contributing area associated with a link. The stream network database consists of a total of about 400,000 links and about 100 environmental predictor variables.

Using presence-absence data for 17 native and alien fish species and environmental attributes derived from the GIS stream network database, Chee & Elith (Ecography, *in review*) were able to develop SDMs with useful predictive ability and discriminatory power. Critically, predictors of distribution that were identified as important for the various species modeled were ecologically interpretable. Resultant SDMs and predictors available for streams across the study area (which encompassed all the inland-draining catchments of Victoria) can be used to provide spatially explicit predictions of species' probabilities of occurrence anywhere within the study area. In other words, to map fish biodiversity patterns across the 127,500 km^2 study area. These methods could be applied to the development of spatial models for other freshwater taxa such as amphibian, macroinvertebrate and freshwater-dependent vegetation species and invertebrate communities.

The GIS stream network database constitutes primary infrastructure for the characterization, representation and modeling analysis of the multidimensional environmental space of freshwater-dependant biota and arguably can be used to support application of the principles of "comprehensiveness" and "representativeness" in Victorian freshwater conservation.

I note that apart from modeling species distributions, it is possible to model the distribution of community-level features such as invertebrate communities, species richness and species turnover. Due to the complexity of community-level modeling and its use in conservation prioritization, this topic will not be further discussed in this report. The reader is referred to Ferrier *et al*. (2009) for further details.

3.2.3 Characterizing Candidate Planning Units

After modeling and mapping the distribution of biodiversity features (i.e individual species, communities, habitat types), this spatial information must be overlaid and integrated to produce an overall understanding of biodiversity patterns across the focal region. Additional characteristics/factors must also be considered in order to identify and select priority areas. I note that the task of identifying priority planning units is first and foremost, a question of assigning and assessing biodiversity and conservation value. The task of selection on the other hand, is embedded within the broader context of the social-ecological system and involves factors that arise largely as a result of alternative potential uses/management actions with respect to the planning unit. These lead on to consideration of practical issues such as vulnerability to threatening processes, cost and feasibility of implementing intended conservation action(s) (see § 3.2.4).

Table 4 lists and describes the purpose or underlying rationale for a range of commonly accepted, widely-used factors. These factors are often used to design criteria that can be used to aid in identification and subsequent selection of priority areas. There are obvious as well as subtle relationships and dynamic interactions between factors relevant to both tasks. For instance, 'naturalness' ('intactness') can be both a conservation value as well as a potential indicator of vulnerability to threatening processes. Areas that are mostly natural/intact may be less vulnerable to threatening processes because their self-regulatory or stabilizing processes are intact and functional. Or they may be less vulnerable to threatening processes simply because they are remote or inaccessible and their 'naturalness' is essentially a reflection of freedom from anthropogenic degradation as a result of remoteness/inaccessibility. Furthermore, remoteness is likely to be associated fewer competitive uses and therefore lower costs of acquisition (assuming the intended conservation action is reservation).

To avoid confusion and the conflating of issues however, it is best to clearly separate the task of identifying priority areas (i.e. assigning and assessing biodiversity and conservation value) from practical concerns that need to be taken into account in the selection process.

Table 4. Commonly used criteria for assigning biodiversity and conservation value (for identifying priority areas) and for assessing practical concerns (when selecting priority areas). Synthesized from Dunn (2003), Nevill & Phillips (2004), Darwall & Vié (2005), Kingsford *et al*. (2005) and Regan *et al*. (2007).

Criteria	Description and explanation of underlying purpose and/or rationale
Biodiversity Value	
Species/community richness	the number of species/communities (whichever is relevant) within a planning unit. The greater the richness, the greater the value of the planning unit.
Species/community/habitat diversity	the full variety of species/communities/habitats (whichever is relevant) within a planning unit. The greater the diversity, the greater the value of the planning unit.
Species aggregation	site/planning unit regularly hosts and/or supports large numbers of species (particularly migratory species)
Significant population numbers	site/planning unit supports a significant proportion of the individuals of a native species
Conservation Value	
Conservation status	the presence-absence or number of species, populations, communities or habitat types that are threatened or endangered. The greater the number of such biological entities in the planning unit, the greater the value of planning unit
Rarity, uniqueness, irreplaceability	The rarity, uniqueness, irreplaceability of species, populations, communities or habitat types within the focal region. The rarer the biological entities in the planning unit, or the more rare entities there are within the planning unit, the greater the value of the planning unit
'Naturalness', 'intactness'	The terms 'naturalness' and 'intactness' are often used to describe ecological condition. The commonality in definitions of 'naturalness'/'intactness' is freedom from anthropogenic degradation and disturbances such as urbanization, clearing, intensive agriculture, grazing, timber harvesting, plantations, mining, extractive industries, water storages, water diversions and river and road engineering works (LCC 1991; Stein *et al*. 1998; Kingsford *et al*. 2005; Linke *et al*. 2007; Thieme *et al*. 2007; Hodgson *et al*. 2009; Nel *et al*. 2009). The greater the degree of 'naturalness'/'intactness', the more valuable the planning unit
Spatial attributes & Landscape context	In this report, spatial attributes refers to characteristics such as the size, shape, orientation, spatial configuration and juxtaposition of planning units. These spatial features have a bearing on population processes, susceptibility to degradation/disturbance and species persistence. See § 3.2.3.1 The landscape context criteria is concerned with properties that arise as a function of a planning unit's landscape context/position that means it plays a role in providing or supporting ecological processes, particularly processes that maintain species populations (see below). Connectivity – does the planning unit provide linkage/movement corridors between areas important as refuges (during periods of environmental stress or natural disturbances) or areas important for fulfilling for species life-history requirements (e.g. mating,

	spawning and nursery grounds)?
	Buffering – planning unit not necessarily important in and of itself, but effectively buffers important areas from adverse influences
	Component within a network of areas - planning unit not necessarily important in and of itself, but its value derives from it being a component in a network with a role in population processes such as facilitating recolonization following local extinction. See § 3.2.3.1
Representation	Number of examples (occurrence level) of the focal biodiversity feature (i.e. species, communities, habitat type, ecological process) within a single planning unit or network of units. The more under-represented the biodiversity feature the more valuable the planning unit
Practical Considerations	
Vulnerability to threatening processes	Risk of future degradation or conversion to production lands, urban development, or for any other purpose that would be detrimental biodiversity within the relevant time frame. See § 3.2.3.2

The conservation value that is assigned to criteria such as 'conservation status' and 'rarity, uniqueness, irreplaceability' is mainly a matter of social judgement and preference. (Afterall, there is no scientific reason *per se* for why a rare species should be of greater value than a common species). However, 'naturalness'/'intactness', representation and the various sub-criteria for spatial attributes and landscape context serve a functional role as surrogates/correlates for characteristics and processes that influence adequacy and persistence. I explicate these relationships in the following sections.

3.2.3.1 Surrogates for Adequacy and Persistence

The ultimate goal of conservation planning is the design of systems that enable biodiversity to persist in the face of natural and human-induced change (Margules & Pressey 2000; Cowling & Pressey 2001). In the research literature, the goal of persistence is often translated into the objective of minimizing extinction risk (Margules & Pressey 2000; Cabeza & Moilanen 2001; Nicholson & Possingham 2006, 2007; Nicholson *et al*. 2006). So why is this not the explicit and direct focus of all quantitative conservation prioritization methods? The short answer is that the evaluation of extinction risk which incorporates an inherent temporal dimension is complicated and resource intensive (Moilanen *et al*. 2009b)

As mentioned in § 3.1, a scientifically rigorous approach to determining adequacy for the purpose of ensuring persistence is a highly complex, multifaceted task that requires grappling with:

a) the specific characteristics of the biological entity (i.e. species, population, community) such as distribution, abundance, habitat and life-history requirements, reproductive capacity, rates of survival and establishment and dispersal abilities;
b) the identification of particular ecological functions and processes that are necessary to sustain/fulfill any aspect of (a);
c) dynamic threats/threatening processes acting either directly or indirectly (such as in the upstream-downstream watersheds or in surrounding regions adjacent to channels and floodplains) on any aspect of (a) and (b); and perhaps
d) conservation actions taken to ameliorate (c)

The points above imply a need for:

i) detailed ecological knowledge of the target biological entity
ii) spatially explicit estimates of vital demographic rates
iii) detailed, spatially explicit knowledge of direct (and indirect) threatening processes and their expected effects
iv) detailed, spatially explicit knowledge of the expected effect of conservation actions on i) to iii).

These data then need to be integrated into models to evaluate the distribution of a species as a dynamic (stochastic) entity, and the time-varying predicted distribution must be translated into an extinction risk using techniques such as metapopulation modeling or spatially explicit population viability modeling (Moilanen *et al*. 2009b). The translation from abundance or distribution to extinction rates requires computationally difficult calculations and specific data about how the spatial pattern or occurrence translates to extinction risk (Moilanen *et al*. 2009b). For instance, habitat quality where a species occurs, influences extinction risk, but habitat quality at locations where the species does not occur could nevertheless influence extinction risk via modified dispersal success and migration mortality (Hanski 1998). Newbold & Siikamäki (2009) provide a recent example of an integrated analysis that incorporates habitat quality models, stock-recruitment and population viability models and economic cost estimates of watershed protection for three closely related Chinook salmon stocks in the Upper Columbia

River basin in Washington. The level of resources required for such an approach means it may only be feasible for a small number of highly valued, threatened or endangered biodiversity features.

As Moilanen *et al*. (2009b) explain, theoretical and empirical observations verify that for most species there would be a threshold effect such that if the quantity and density of suitable habitat falls below a particular threshold, the metapopulation is likely to become unviable and spiral to extinction. Unfortunately, the identification of thresholds for any real-world species and system is complex.

Lack of data and specialist expertise, uncertainty of model components and parameters and computational limitations are contributing reasons to why direct minimization of extinction rates is an uncommon goal in multi-species conservation (Moilanen *et al*. 2009b). Instead, the approaches to minimizing extinction rates are typically indirect and rely on factors that promote species persistence through conservation/management of spatial areas and features that sustain ecological functions and processes (see § 2.1.1) and foster resilience and evolutionary (adaptive) potential (Cowling *et al*. 1999; Cowling & Pressey 2001; Desmet *et al*. 2002; Rouget *et al*. 2003; Moilanen *et al*. 2008; Roux *et al*. 2008; Klein *et al*. 2009). There are two fundamental guiding principles with theoretical and empirical support: firstly, with all things being equal, species are more likely to persist in suitable rather than unsuitable habitats and secondly, large, compact (lower edge to area ratio), aggregated and better connected areas are better than smaller, scattered ones (Araújo 2009). Common surrogates for adequacy and persistence include:

a) ecological 'condition' (e.g. Nel *et al*. 2009) and/or some dimension of habitat quality (.e.g. water availability and quality during the driest quarter in a year, Roux *et al*. 2008),
b) spatial attributes or arrangements that support population and ecological processes, particularly by facilitating connectivity between areas with different resources and different areas important for completing life-history stages (Moilanen *et al*. 2008);
c) spatial attributes or arrangements that support adaptive and evolutionary potential (Cowling *et al*. 1999; Cowling & Pressey 2001; Desmet *et al*. 2002; Rouget *et al*. 2003)
d) spatial attributes or arrangements that buffer against adverse effects (Regan *et al*. 2007; Klein *et al*. 2009); and
e) higher levels of representation of focal community/habitat types or viable populations of focal species (e.g. Roux *et al*. 2008) to provide insurance and risk-spreading (Commonwealth of Australia 1997; Moilanen *et al*. 2009b; Palmer *et al*. 2009).

'Naturalness'/'Intactness'

'Naturalness'/'intactness' is, for practical purposes, frequently defined as the freedom from anthropogenic degradation and is often used to indicate ecological 'condition' (e.g. Linke *et al*. 2007; Nel *et al*. 2009) or habitat quality (e.g. Hodgson *et al*. 2009). With respect to 'condition', the expectation is that areas/catchments that are mostly natural or intact retain their ecological integrity – their powerful self regulatory and self-stabilizing processes (Ricklefs *et al*. 1984), which enables "an ecosystem to continue its natural path of evolution, its normal transition over time, and its successional recovery from disturbances" (Westra *et al*. 2000, cited in Nel *et al*. 2009). Habitat quality or suitability, as mentioned above, is important for sustaining viable population sizes and growth.

'Naturalness'/'intactness' implies an absence of anthropogenic modifications to any natural feature(s) in the catchment that might impinge on hydrological, geomorphic and ecological functioning and processes. Given such a broad perspective, there are clearly myriad ways to construct estimates of 'naturalness' and the particular details would depend on user-requirements. One of the most detailed analyses of 'naturalness' in riverine contexts, is Stein *et* *al*.'s (1998, 2002) spatial analysis of anthropogenic river disturbance at continental and regional scales in Australia.

Stein *et al*.'s (1998, 2002) scheme used a continuous scale for rating disturbance from severely degraded to near-pristine. Multiple surrogate measures were used to link the impact of human activities with the stream via overland flow within the catchment and via stream topology to derive empirical indices of river disturbance. Their method took into account a) both point and diffuse impacts on water quality and b) both in-stream (e.g. impoundments, diversions) and catchment (e.g. settlement, landuse) factors that alter flow regimes. At the continental scale, more than 1.5 million stream links (i.e. the portion of a stream between junctions) with a total length of over 3 million km were assessed and rated to produce a comprehensive and nationally-consistent characterization of river and catchment disturbance (Stein *et al*. 1998, 2002).

Spatial Attributes, Landscape Context & Representation

Spatial attributes such as size, shape, orientation, spatial configuration and juxtaposition of planning units have a bearing on population processes (e.g. carrying capacity and movement patterns), susceptibility to degradation/disturbance (e.g. edge effects and spatially-correlated disturbances such as fire and diseases) and ultimately, species persistence. Table 5 summarizes the ecological rationale and arguments for dominant guidelines on spatial attributes.

Table 5. Explanation of the importance of various spatial attributes for ensuring biodiversity persistence.

Spatial Attribute	Ecological rationale
Size	Large areas are likely to enhance the persistence of biodiversity features because they tend to contain greater amounts of particular habitat types and cover a matrix of different successional stages and alternative stable states, and more populations, more individuals, and greater genetic diversity of individual species (Gaston *et al*. 2008). The assumption above is that the habitat types are of adequate/suitable quality for sustaining the individuals and populations therein. Large areas are also expected to ensure continuation of large scale ecological processes such as long-distance migration (e.g. of diadromous fish species in freshwater environments) and natural regimes of disturbance and recovery (e.g. overbank flooding, fire, drought). With large, heterogeneous areas, at least some habitats and populations can be expected to escape, survive or recover from large-scale stochastic disturbances or threatening processes (Opdam & Wascher 2004).
Shape	Compact areas have a lower edge-to-area ratio. This reduces 'edge effects' which may enhance local persistence (Cabeza & Moilanen 2001). Compact areas are also easier and less costly to manage than highly fragmented and/or widely distributed areas (Cabeza & Moilanen 2001; Moilanen & Wintle 2006).
Configuration, arrangement, juxtaposition	Clustered, aggregated configurations enhance connectivity and facilitate dispersal, recolonization of unoccupied habitat (Cabeza & Moilanen 2001) and movement of the various life-history stages of organisms, all of which can contribute towards local and regional persistence. Shorter dispersal distances may also reduce dispersal mortality. The flip side to clustered arrangements is that it increases the risk of simultaneous extinction in the face of large-scale, spatially correlated or contagious disturbances such as cyclones, wildfire and pathogens (Cabeza & Moilanen 2001; Araújo 2009).

The recommendation for 'compactness' is obviously of limited application in riverine systems which are naturally longitudinal, distributive networks that are generally channel constrained (Moilanen *et al*. 2008).

As described in § 2.1 and 3.2.2.3, riverine ecosystems comprise complex longitudinal and lateral mosaics of habitat patches at multiple temporal scales. This means that elements that are important as refuges (during periods of environmental stress or natural disturbances) or are critical for the different life history stages of freshwater biota are often separated in space and time. This is true particularly with respect to arid, drought-prone river systems in the first instance, and diadromous fish species in the second instance, where mating, spawning and nursery habitats may be geographically distant from the species' regular habitat. Landscape context is therefore of critical importance and interest. As outlined in Table 4, there are a number of dimensions to landscape context and its influence on connectivity and buffering against adverse impacts within a reserve network design.

Connectivity is an important consideration, because it influences dispersal, colonization and population sizes at sites. According to a fundamental tenet of spatial (meta)population dynamics, of two sites with otherwise identical features, the one with higher connectivity would have higher population densities (Hanski 1998). Many different connectivity measures and indices have been developed to quantify and assess the connectivity of different sites (see e.g. Moilanen & Nieminen 2002; Matisziw & Murray 2009). Many of these connectivity measures/indices were developed for terrestrial and marine ecosystems and are not directly transferable to riverine ecosystems. For instance, many measures are based on patch center-to-center distances for nearest neighbour measures or circular buffer measures. Connectivity in riverine systems cannot be measured in an analogous manner because it is strongly directional and generally channel constrained (Moilanen *et al*. 2008). In addition, connectivity is mediated by the fluvial medium, so it is important to consider not only upstream-downstream and adjacency connections but also if the flow regime is adequate to ensure hydrologic connectivity.

As mentioned in Table 4, a planning unit can also conservation value if it serves as an effective buffer against adverse influences. This is particularly relevant to riverine ecosystems as the state, condition and activities in the entire upstream contributing catchment area (encompassing the catchment areas of upstream tributaries) can affect environmental conditions at a focal planning unit. Planning units that yields relatively high volumes of runoff may be important to retain as a buffers. Planning units that contain elements that are important during periods of environmental stress or natural disturbances such as drought and thermal refuges also contribute towards conservation value. In a recent study, Klein *et al*. (2009) described how they identified and targeted drought and evolutionary refugia in a continental-scale conservation planning exercise.

Representation refers to the amount or number of instances of the focal feature, which could be a particular community or habitat type or populations of a particular species. Representation serves as a criteria for assessing conservation value, but also has a bearing on biodiversity persistence. In Table 5, the potential risk of clustering was noted. Setting high levels of representation within a focal region or requiring representation across a wide range of geographic, environmental and biotic domains are useful risk-spreading strategies that can help to insure against the risk of simultaneous extinction in the event of large-scale spatially-correlated stochastic disturbance. Similar features in different locations may also be differentially impacted by climate change because of their particular local context (e.g. wetland habitat in a catchment with negligible water resource and agricultural development versus wetland habitat in heavily developed catchment) (Palmer *et al*. 2009).

3.2.3.2 Vulnerability to Threatening Processes

Pressey *et al*. (1994) defined vulnerability as the likelihood or imminence of biodiversity loss caused by current or impending threatening processes. Wilson *et al*. (2005) expanded the concept by outlining three dimensions to vulnerability:

a) exposure - the probability of a threatening process affecting an area over a specified time or the expected time until an area is affected
b) intensity - magnitude, frequency and duration of a threatening process
c) impact - the response of species or other biodiversity features to the threat

A comprehensive, rigorous approach to quantifying vulnerability has much in common with Ecological Risk Assessment (see Burgman 2005) and would, at a minimum, involve: threat identification and assessment of exposure, intensity and impact of threats acting individually or in combination. The threatening processes listed in § 2.2 provide a good starting point for the former task and a wide range of inductive techniques are available to assist with systematic threat identification and detailed characterization of their potential effects (see e.g. Carey *et al*. 2004; Burgman 2005). Ideally, it would be valuable to map the patterns of individual threats and monitor their rates of spread. Assessment of exposure, intensity and impact of threats typically requires quantitative tools and models as well as methods to account for uncertainty (see Burgman 2005 for further details).

Exposure alone however, is what is most commonly dealt with in spatial conservation prioritization. This reflects the difficulty of empirically estimating (parametrizing) intensity, impact and the effects of their interaction (Possingham *et al*. 2009). Nevertheless, even some fairly basic information on vulnerability can assist with selecting priority areas. For instance, it is useful to distinguish between 'stoppable' and 'unstoppable' threats to a planning unit. If a threat cannot be abated/mitigated and/or is likely to cause impacts that are extremely resource-intensive to rehabilitate (salinization of a region underlain by saline groundwater aquifers is an example of such a threat), then the planning unit should be avoided.

Information on vulnerability can also assist with scheduling conservation actions. For example, at the conclusion of a spatial conservation prioritization assessment, it is rarely possibly to execute conservation actions on all priority areas instantaneously. During the intervening period in which the conservation plan is implemented incrementally, threatening processes may continue to operate and consequently alter the conservation value of high-priority areas (Cowling *et al*. 1999). Information on vulnerability can partially account for this to ensure conservation effectiveness and efficiency when conservation actions must be staged/scheduled over many years (Dreschler 2005; Moilanen & Cabeza 2007). This information is of course, most useful if it is spatially explicit.

Climate change poses a potentially pervasive threat. Climate change raises the prospect of higher temperatures, reduced rainfall, increased evaporation, reduced soil moisture levels, runoff, streamflow and groundwater recharge. This profound set of spatially diffuse changes is likely to engender complex, cascading effects on components of the hydrological cycle (and ultimately the natural flow regime), carbon/nutrient inputs and rates of cycling, natural disturbance regimes (such as droughts, floods and fire), biological interactions (such as competitive relations, invasions and pathogen loads) and water and land use dynamics (such as increased extractive uses and land conversion) as growing human populations adapt to the complex challenges of climate change. In turn, these changes and the synergistic interactions amongst them will influence individual species' responses and aggregate patterns of biodiversity. However, the nature and trajectories of these changes is poorly understood and highly uncertain. To date, the implications of these dynamic threats for freshwater biodiversity have received little attention. This underlines the urgent need to develop strategic capabilities for anticipating, monitoring and responding to dynamic and emergent threats.

3.2.4 Constraints & Conservation Actions

Constraints limit the set of permissible conservation actions. Constraints come in many forms and can be budgetary, social, political, institutional, legal and regulatory in nature (Moilanen *et al*. 2009b). This section concentrates on budget constraints, in large part because they are concrete and relatively easy to define. However, I fully acknowledge the critical importance of other forms of constraints that may be less tangible.

Ferrier & Wintle (2009) noted that in much of the existing literature on 'systematic conservation planning' or conservation prioritization, it is assumed that the action requiring prioritization is the establishment of new conservation reserves (or some form of protection). However, this is a rather more limited view than in real-world situations where the conservation actions requiring (spatial) prioritization could be measures relating to

a) amelioration of threatening processes (e.g. weed control, control of feral predators, removal of riparian grazing pressure and regulation of groundwater extraction);
b) restoration (e.g. re-snagging/re-introducing coarse woody debris into channels to restore habitat processes, reinstating components of the natural flow regime, removal of fish barriers preventing upstream-downstream movement and replanting riparian vegetation); and
c) allocation of funding for conservation management activities (including the two described directly above).

It is important to be clear about the conservation action under consideration because the relative priority of spatial locations for one potential action (e.g. reservation) will almost certainly differ from that for another action (e.g. restoration by re-snagging).

Resource limitations are why we need to prioritize in the first place. Budget constraints limit what we can do (conservation actions) and/or how many planning units we can act on. The total budget determines how many sites/planning units can be reserved or how much area can be controlled for invasive species or restored by replanting. All this requires information on the cost of different actions as well as how these costs might vary across different parts of the landscape. If cost data are sparse, spatial interpolation and modeling may be required to fill gaps in spatial coverage in a manner analogous to that used for biological data.

If the conservation action is protection or acquisition for conservation, then applicable costs include acquisition costs for land or water rights, transaction costs, and costs associated with management and maintenance and opportunity costs (Naidoo *et al*. 2006). Opportunity costs relate to the value of forgone opportunities since the use of an area or water resource for protection and conservation generally closes off alternative potential uses such as timber, forage and agricultural production or even residential development. It is however, important to also recognize that these opportunity costs may be offset by benefits associated with protection or acquisition for conservation, such as improvement of downstream water quality or increase in recreational opportunities. The difficulty of course, is that it is difficult to quantify and render ecosystem service-type benefits comparable to costs in monetary terms. Nevertheless, realistic accounting of cost estimates for conservation actions is important for ensuring credibility and cost-effectiveness (discussed below) in the prioritization process.

Cost-effectiveness implies that available resources must be used for maximum conservation benefit, or alternatively, that pre-defined conservation objectives should be satisfied with minimal cost (Moilanen 2008). Either way, gauging cost-effectiveness for a given resource level requires (spatially explicit) estimates of costs as well as ecological benefits of conservation actions. Estimates of benefits should ideally be in a form directly related to the performance measure for the conservation objective. For instance, how many species are conserved if a planning unit is chosen as a reserve or invasive species are controlled or habitat restored. Estimating how many species are conserved if a planning unit is chosen as a reserve is relatively straightforward and is a routinely done in conservation prioritization studies (see e.g. Moilanen *et al*. 2008 for an example in a freshwater environment). Estimating benefits associated with ameliorating threats or restoring habitats is challenging, as evident from the few examples available in the literature (e.g. Thomson *et al*. 2009). In essence, the question being asked is, how does the choice of action(s) influence the distribution or abundance or extinction risk of focal biological entities? This is generally tackled using one or more often, a suite of inter-linked statistical, landscape and population dynamics models (see e.g. Newbold & Siikamäki 2009; Thomson *et al*. 2009). Given the range of potential processes involved in conservation actions associated with ameliorating threats or restoring habitats and the specific characteristics of different biological entities, ready-made models and assessment frameworks are unlikely to be available and will have to be developed on a case-by-case basis.

In the literature, the problem most commonly analyzed is spatial allocation of a single conservation action (which is usually choosing planning units for reservation). Many real-world situations however, are not so simple. Even with the choice of a single action, planning units could be allocated varying levels of protection (see e.g. the various IUCN Protected Area Categories), threat amelioration and restoration. In other words, the effects (expected benefits) of a single conservation action can span a continuum. In reality too, multiple actions are fairly common. For example the suite of riverine-related management actions in a catchment often include fencing exclusion of riparian zones, control of weed species, riparian replanting and environmental flow management.

As Moilanen *et al*. (2009b) explain, multi-action planning is structurally much more complicated than single-action planning because there may be trade-offs between features and non-independence between actions. An action that is beneficial for one feature might be less beneficial or perhaps even harmful for another feature (van Teeffelen & Moilanen 2008). The influence of an action at one site may influence features in another site and the combined impact of multiple actions may differ from their sole effect. For instance, removal of instream fish barriers may benefit certain fish species that move long-distances to complete life-history requirements, but it may also expose species in previously inaccessible rivers and streams to predatory pressures from alien species. The potential for positive, negative and synergistic interactions means multi-action conservation planning can be a complicated, non-linear optimization problem. Having multiple actions makes both the performance evaluation of a candidate solution and the search for solutions more complicated and computationally more demanding (Moilanen *et al*. 2009).

3.2.5 Selection Strategies & Optimization

To begin with, I give a brief overview of two fairly commonly-used, semi-quantitative approaches to identifying and selecting priority areas for conservation and indicate their limitations. I then shift the focus to quantitative approaches which provide a more rigorous treatment of the task, which is to select from amongst all candidate planning units, a set that will give the best outcome (according to the performance measure), whilst taking into account the conservation action(s), constraints and various considerations regarding persistence and threatening processes. This is a complex job and performance evaluation of candidate solutions requires mathematical and computational tools.

3.2.5.1 Criterion-based approach

International programs or frameworks for identification of places of conservation value (such as World Heritage Convention and Ramsar Convention) are often based on a criterion-based approach. In this approach, explicit criteria are defined, often along with specific (qualitative and/or quantitative) thresholds for particular criteria and decision rules. Selection is based on satisfaction of criteria at the agreed standard. The principal feature of this approach is that each site is evaluated only against the criteria and not against other sites of the same/comparable type, so there is no limit to the number of sites of particular types (Darwall & Vié 2005; Kingsford *et al*. 2005). The extent to which considerations such as representativeness, persistence and threatening processes are addressed depends entirely on how the criteria are constructed (for instance, species-specific criteria aimed at promoting persistence may be defined). There are typically no mechanisms for accommodating constraints, so a separate process will be required for prioritization in many real-world situations.

3.2.5.2 Scoring approaches

With scoring methods, candidate planning units evaluated individually and independently of one another against some pre-defined set of criteria that are intended to capture/reflect 'value' (e.g. the criteria in Table 4). Resultant scores are used to create a ranked list, from which planning units are then selected.

Scoring schemes can accommodate large numbers of disparate, non-commensurate factors. For example, the list of factors considered by the Rivers and Streams Special Investigation included botanical and faunal qualities, presence of endangered native fish species, geological features, 'naturalness', scenic, cultural heritage and recreational qualities. Scoring scales can be tailored to be categorical, ordinal or numeric, as required for individual factors. The assignment of scores for individual factors can be qualitative, based on subjective judgements or based on objective quantitative data. If different scoring scales are used then some form of transformation is usually required to standardize scores from different factors. After standardization, scores can either be provided 'as is' or aggregated in some manner to generate a composite index. Composite indices are usually constructed using a simple, additive method, often with weights applied to the factors. The establishment of weights for the different factors may be based on the subjective judgements of the developers of the scoring approach or on the opinions of a wider group of individuals (or experts). When the weighting is to be determined by a group of individuals, the process is often supported by techniques such as the Delphi technique (Burgman 2005) and other techniques from the field of multi-criteria decision making such as the Analytical Hierarchy Process (Regan *et al*. 2007). A ranked list is then generated from the composite index.

Scoring approaches are widely used, probably because they are relatively easy to develop, implement and describe (Ferrier & Wintle 2009). Filipe *et al*. (2004) provides an example of its application for selecting priority areas for fish conservation in Portugal. It is however, important to understand the limitations of scoring methods.

The use of additive scoring implies that the factors are independent and substitutable (Burgman *et al*. 2001). If factors are not independent (that is to say, they are correlated in some way), then the use of additive scoring implies a degree of 'double-counting' for correlated factors. The fact that scores are added together also implies that one factor is substantively equivalent to any other factor – in the overall scheme, a low score in one factor can be perfectly compensated for by a higher score in another factor. So for example with respect to the list of factors above, use of additive scoring assumes that there is no relationship between botanical and faunal qualities and 'naturalness'. It also implies that a complete loss of 'naturalness' can be perfectly compensated for by a high score for recreational qualities. It is unlikely that this accurately reflects the intent of developers of scoring schemes. Unfortunately, the ability of (additive) scoring schemes to incorporate large numbers of disparate, non-commensurate factors is largely illusory.

Even assuming that the factors used are indeed fully independent and substitutable, scoring may be inefficient with respect to representativeness if it turns out that planning units with the highest scores (i.e. highly-ranked units) are ecologically similar (in the sense that they contain similar biodiversity features such as species, communities or habitat types). Selecting from the top of a ranked list in this scenario might mean missing biological features that only occur in planning units with lower scores (Moilanen *et al*. 2009b). Finally, in scoring approaches, the value of a set/network of planning units is implicitly a sum over the scores of the highest ranked planning units and this ignores the fact that in inherently interconnected ecosystems, the value of a set is more than simply the sum of its parts.

3.2.5.2 Complementarity-based approaches

Complementarity has been defined in different ways by different researchers and is a potentially confusing concept (e.g. Vane-Wright *et al*. 1991; Margules & Pressey 2000; Cabeza & Moilanen 2001; Margules & Sarkar 2007). Moilanen (2008) recently deconstructed and clarified the concept and in doing so, proposed a generalized notion of complementarity which is the one adopted in this report.

Generalized complementarity has been named the ‘conservation interactions principle’ (Moilanen 2008) to reflect the fact that there may be interactions amongst actions as well as spatial dependencies, which can influence trade-offs between features (as described in § 3.2.4). The fundamental point is that “conservation benefits of all conservation actions across the landscape should be evaluated jointly and account for long-term consequences of interactions between actions” (Moilanen 2008). As he explains, the conservation interactions principle has explicit mathematical and technical meaning, and it implies that conservation benefits that follow from a particular conservation action at a site depend on the regional context of the site and conservation actions taken elsewhere. Applying this perspective, the performance of each biodiversity feature (i.e species, communities, habitat type, ecological process) is first evaluated jointly across the landscape and then the performance across features is aggregated; thus dependence between site and actions is included (Moilanen 2008). In contrast, scoring approaches first evaluate performance (independently) within each site and then across sites, creating potential duplication of effort.

Another point of difference between quantitative complementarity-based approaches and criterion-based or scoring approaches is rigorous performance evaluation. Candidate solutions are evaluated using pre-defined performance measures and this provides a transparent, substantive basis for comparing alternative solutions. For instance, if the performance measure is number of species conserved, this approach allows an analyst to make statements such as, choosing priority areas in set X conserves 50% more species than choosing set Y. As mentioned previously, this is a complex task when it involves multiple species, consideration of various surrogate measures of persistence, connectivity, threats and constraints and is therefore reliant on mathematical and computational tools. These tools, often referred to as site selection algorithms, have been implemented in numerous conservation planning software tools. Prominent examples from the academic literature include Zonation, Marxan and C-Plan. Each software package also includes variants of the core algorithms for different purposes and emphases. The reader is referred to Moilanen *et al*. (2009a) and chapters therein for details of the various packages, their features, capabilities and limitations.

In the following section I briefly describe two major strategies for selection: target-based and benefit function-based. Target-based planning is perhaps the most widespread selection strategy. Targets are specified for the desired representation levels of each biodiversity feature. If the biodiversity feature is a species, the target could be some minimum number of point occurrences, or populations. If the biodiversity feature is a habitat type, the target could be a minimum area or perhaps a fixed proportion (e.g. 10%) of say, the total area of the habitat type (prior to any loss due to habitat clearance or transformation). Using the guiding principle of cost-effectiveness, selection algorithms for target-based planning aim to either:

a) identify sets of planning units that meet defined targets as cost-effectively as possible (known as the minimum-set coverage problem); or
b) maximize the number of targets achieved for a fixed resource level (known as the maximum coverage problem)

The output of these target-based methods is usually a single optimal set of sites and one or a small number of the best solutions found. Linke *et al*. (2007), Roux *et al*. (2008) and Amis *et al*. (2009) provide example applications in Victorian and South African freshwater environments. Analyses by Linke *et al*. (2007) and Amis *et al*. (2009) were conducted using Marxan software (with modifications) and the study by Roux *et al*. (2008) employed C-Plan software.

Targets provide a clear goal to aim for, are convenient and easy to communicate (Margules & Pressey 2000; Svancara *et al*. 2005). However, they are relatively arbitrary and are not necessarily reliable surrogates for species persistence (Cabeza & Moilanen 2003; Svancara *et al*. 2005). To address the issue of arbitrariness, numerous schemes have been developed for assigning varying proportional targets to different biodiversity features based on explicit consideration of attributes such as the natural rarity of each feature, past depletion, vulnerability to threats, and internal biological richness and heterogeneity (Ferrier *et al*. 2009). As Ferrier *et al*. (2009) explains, these schemes typically either

a) categorize features according to once or more of these attributes, and then assign fixed proportional targets to each of the categories - for example, 15% of original area for widespread non-vulnerable classes, 60% of remaining area for vulnerable classes, and 100% of remaining area for rare and endangered classes (Commonwealth of Australia 1997); or
b) apply some form of upscaling of targets as a continuous function of varying levels of rarity, depletion, vulnerability or heterogeneity.

These schemes are arguably still arbitrary, so various theoretical methods have been developed for setting or justifying targets (see Ferrier *et al*. 2009 for details). In the last couple of decades, much effort has been expended on refining and adjusting the target-based paradigm to reflect the subtle complexities of real-world situations. Adaptations include changes in the way targets are defined to account for rarity, depletion and vulnerability (as discussed above), defining targets for retention or restoration of habitat rather than just protection, and assigning targets to areas required for the maintenance of ecological or evolutionary processes (e.g. Klein *et al*. 2009) (Ferrier *et al*. 2009).

Benefit functions represent a different conception of the selection problem. In the benefit function framework the objective is to maximize the value of the full set or network, however value is defined (Moilanen *et al*. 2005; Moilanen 2007). The Zonation framework and software effectively addresses a maximum utility type problem whilst explicitly taking into account the distributions of biodiversity features, considerations of connectivity, constraints and uncertainty (Moilanen *et al*. 2009c). The following brief description of Zonation is drawn from Moilanen *et al*. (2009c). The Zonation algorithm does not require the specification of biodiversity feature targets - it starts with the full landscape and iteratively removes selection units whose loss causes the smallest marginal loss in the overall conservation value of the remaining landscape. The result is a hierarchy of conservation priority through the landscape which can be visualized as a colour-coded map. In this single nested hierarchy, all selection units are ranked with the best 1% of the landscape nested within the best 2%, which is in turn nested within the best 3% and so on, providing a continuous representation of priority. The performance of individual biodiversity features at different levels of landscape removal can also be viewed. Zonation includes sophisticated tools for handling species specific connectivity responses as well as various definitions of marginal loss. In conjunction, these capabilities enhance the power and flexibility of Zonation for addressing subtle shadings in conservation prioritization problems. Moilanen *et al*. (2008) provide an example application of riverine conservation prioritization in New Zealand with realistic modeling of directional connectivity.

3.3 Applying spatial freshwater conservation prioritization in Victoria

§ 3.2 describes an 'operational model' (*sensu* Knight *et al*. 2006) of how a spatial conservation prioritization process functions. The recent Victorian Government White Paper on Land and Biodiversity in a Time of Climate Change (DSE 2009) refers, in Action 6.3.3 (p.91) to updating the existing prioritization system to identify high conservation value aquatic ecosystems by 2011. It would be interesting to compare this existing prioritization system with the spatial conservation prioritization process described in § 3.2.

The recently developed quantitative methods and spatial tools described in § 3.2.2.3 have the potential to transform and significantly improve prioritization and decision-making processes. Essentially, the GIS stream network database and species distribution models (SDMs) enable spatially explicit, ecologically interpretable predictions of biodiversity distribution patterns in river networks for any region of interest in Victoria. These predictions can be visualised at fine-scales and the spatial models constitute the basic inputs for spatial conservation prioritization using sophisticated optimization tools. A data-driven, quantitative approach to prioritization confers the following advantages:

- Explicit and therefore repeatable and auditable
- Use of a mathematical technique enforces a degree of rigour in problem formulation and all intermediary steps towards implementation. For instance, with respect to the specification of high-order goals, means objectives and constraints.
- Features 1 & 2 encompass documentation processes and contribute to the construction of a tangible knowledge base for the problem
- Enhances scientific credibility
- Enables the prioritization process to be linked to monitoring, evaluation and reporting of progress towards achieving conservation goals (Ferrier & Wintle 2009)
- In theory it enables the prioritization process to be linked to processes that address influences on biodiversity persistence (e.g. dispersal and connectivity; metapopulation dynamics; population viability analyses etc) and allows these to be taken into account.

The data, scientific tools and methods exist to aid freshwater conservation prioritization, but in real-world planning and decision-making involving multiple actors across multiple jurisdictions (many of whom may have competing interests) much effort will be required to formulate clear goals, objectives, conservation actions and constraints.

4 SUPPORTING EFFECTIVE FRESHWATER CONSERVATION

It is useful to distinguish between three broad phases of conservation practice, namely systematic assessment, planning, and implementation and management. Although each phase is conceptually and methodologically separate, they are clearly closely interrelated and a sound understanding the interplay between these phases is necessary for effective freshwater conservation. § 3.2 described an operational model for systematic assessment. In this section, I discuss recommendations and issues that relate to the planning, implementation and management stages.

Two basic premises underlie the recommendations in this section. They are that water resource development, river regulation and land management practices have already wrought profound changes on natural flow regimes and that climate change scenarios should be incorporated into all planning and action. Heller & Zavaleta (2009) concluded from their review of 22 years of recommendations on biodiversity management in the face of climate change that the majority of recommendations in the published journal literature are too generic and lack sufficient specificity to direct action. I have therefore endeavoured as far as possible, to describe relevant recommendations for freshwater-dependant ecosystems in actionable terms.

Consistent recommendations from the academic literature for strategies that are likely to be robust to unavoidable uncertainty about climate change and future conditions can be broadly categorized under the following headings:

a) Protected areas
b) Sympathetic whole-of-catchment management
c) Connectivity
d) Restoration and management
e) Reduce threats and pressures
f) Refugia
g) Translocation and reintroduction

The implicit function of these recommendations is to foster biodiversity and ecosystem resilience. Resilience is defined as the capacity of a system to maintain its characteristic patterns, structures, functions and rates of processes (such as primary productivity, energy exchange, nutrient cycling and food-web structure) despite perturbations (Walker 1992). Resilience derives from partially redundant control processes that act at different scales to mitigate effects of perturbations (Carpenter & Cottingham 1997). Although there is some degree of in-built resilience in the functioning of ecosystems, usually it is difficult to determine the exact basis, boundaries or limits of this resilience. The ecological rationale (the 'what' and the 'why') of these recommendations are described in the following sections. Some of these extend across more than one category. Wherever possible, I have attempted to also outline the 'how' and 'by whom'.

4.1 Protected areas

Numerous authors have advocated protecting large areas, increasing the size and scale of protected areas and ensuring that network of protected areas incorporates regions of high environmental/habitat heterogeneity (Saunders *et al*. 2002; Opdam & Wascher 2004; Kingsford *et al*. 2005; Kingsford & Nevill 2006; Hannah *et al*. 2007; Heino *et al*. 2009; Hodgson *et al*. 2009; Ormerod 2009; Palmer *et al*. 2009).

The following insights from evolutionary conservation biology are salient (Holt & Gomulkiewicz 2004):

a) populations that are initially rare are highly vulnerable to even moderate environmental change
b) even large population are vulnerable to strong environmental change
c) given sufficient variation to adapt to abrupt environmental change, in a deterministic world, a population should eventually bounce back from low numbers (i.e. evolutionary rescue is facilitated by generic variability). But when numbers become too low, even well-adapted populations may face extinction from demographic stochasticity
v) if the environment changes sufficiently fast so that a species in its initial state reaches low densities over short time scales (e.g. tens of generations), natural selection is unlikely to be effective at preventing extinction.

As Holt & Gomulkiewicz (2004) point out, the theory suggests two distinct avenues through which it might be possible to foster conservation and species persistence:

1. Population size is the product of area and density. Total carrying capacity always steadily increases with increasing habitat area and quality (which is a proxy for density). Populations in large areas take longer to decline to a given absolute abundance than do populations in small areas. These points, along with the points in Table 5 provide strong justifications for increasing the size, scale and habitat quality of protected areas.

2. If the rate of decline can be slowed, populations have an enhanced "window of opportunity" in which to evolve adaptations to environmental stresses. So, if we cannot prevent environmental change, we may be able to reduce the magnitude of its impact upon a focal species. This lengthens the time scale available for evolutionary change and provides more opportunity for evolution by natural selection to alter the species' niche sufficiently to ensure persistence in the novel environment.

Niche conservatism is the phenomenon in which species seems to exhibit much the same ecological niche (defined as the range of environmental conditions, resources, etc. needed for population persistence) over its geographical ranges and over evolutionary time scales (even during epochs of massive changes in environmental conditions (Holt & Barfield 2008). The implication is that such species will only be able to survive changing conditions through time if they can track shifts in their ecological niche through geographic space.

The argument for incorporating areas with high environmental/habitat heterogeneity such as areas with steep elevation and climatic gradients is that they provide greater opportunities for populations to survive by tracking or shifting between habitats with suitable environmental conditions (Heino *et al*. 2009; Hodgson *et al*. 2009; Palmer *et al*. 2009). As mentioned in Table 5, large, heterogeneous areas, provide some buffer against vulnerability because at least some parts and populations can be expected to escape, survive or recover from large-scale stochastic disturbances or threatening processes. A network of protected areas consisting of source and sink habitats is important for maintaining population processes (see Table 4). Source areas should be prioritized before sink areas, but sinks have value, particularly with respect to providing opportunity for niche evolution – this could help reshape species niches and allow adaptation to altered environmental conditions *in situ* (Holt & Gomulkiewicz 2004).

Australian freshwater biota have evolved in the evolutionary context of high natural (spatial and temporal) variability of Australian riverine ecosystems. This is reflected in the "richness of refugial strategies is in contrast to the paucity elsewhere" Lake (1995). Hopefully, these factors provide a basis for inherent resilience which will respond to our efforts to enhance it.

In freshwater ecosystems, habitat area, volume, heterogeneity and connectivity is created by the fluvial medium interacting with the geomorphic template. So provision of adequate and appropriate watering/flow regimes is at least as important as river protection with respect to the conservation of instream habitat and species persistence. Any protected area designation for riverine and freshwater-dependent ecosystems must to be accompanied by commitments for adequate and appropriate watering regimes. Failure to ensure appropriate flow regimes will lead to continuing habitat loss/degradation.

The focus of protected area identification, design and management has predominantly on terrestrial biodiversity with freshwater biodiversity often only protected incidentally through coincidence of being incorporated in terrestrial protected areas (Saunders *et al*. 2002; Amis *et al*. 2009; Heino *et al*. 2009; Nel *et al*. 2009. And even then, not necessarily so (see below). There is an urgent need to integrate the conservation of freshwater and terrestrial biodiversity (Abell *et al*. 2007; Amis *et al*. 2009). Failure to jointly assess freshwater and terrestrial biodiversity results in bias towards terrestrial ecosystems and in effect undervalues the linkages between them. Neglecting to account for the biodiversity of a highly valued biome is contrary to the principles of comprehensiveness, adequacy and representativeness. And neglect of terrestrial-freshwater linkages is a missed opportunity for holistic, coherent multi-scale action.

In practical terms, a useful first step may be to increase the level of protection afforded to freshwater ecosystems situated within terrestrial National Parks. This is presently not the case in Victorian National Parks where recreational fishing is permitted (Nevill & Phillips 2004). Following that we might consider conducting a systematic assessment to investigate options for expanding the protected area system (*sensu* Nel *et al*. 2009). Priority should be given to intact areas so that we can capitalize on the powerful self regulatory and self-stabilizing ecological functions and processes of intact or nearly intact ecosystems rather than rely on costly continuous intervention (Ricklefs *et al*. 1984). This of course, may not be possible in regions that have already been substantially transformed and/or are predominantly in private landhold, which brings us to the next section.

4.2 Sympathetic whole-of-catchment management

The importance of sympathetic and judicious whole-of-catchment management for freshwater biodiversity conservation is well-recognized has been long advocated (Pringle 2001; Saunders *et al*. 2002; Heino *et al*. 2009, Nel *et al*. 2009; Palmer *et al*. 2009). Riverine ecosystems are influenced by anthropogenic activities throughout the catchment in which they are situated, via one or more of the interactive pathways along their longitudinal, lateral and vertical dimensions (see § 2.1). In other words, all water resource and land use activities and management in a catchment can potentially impact riverine ecosystems and therefore, matter.

As mentioned in the section above and § 2.2, habitat loss and degradation in Victoria has been most prevalent and intense in areas with landuse potential for agricultural production, natural resource extraction and urban development. Such regions are concentrated around the lower reaches of larger river systems as well as estuaries. About 66% of the state of Victoria is private land (DSE 2003) and over 80% of vegetation cover has been removed from privately held land (DSE 2009d). In situations where intact habitat is relatively scarce, there is nevertheless a continuum extending across near-natural and semi-natural habitats and managing these areas well contributes towards mitigating deleterious effects of landuse conversion. The structure and quality of areas surrounding actual habitat occupied by a species is important because it can influence extinction risk via dispersal success and migration mortality (Hanski 1998; Opdam & Wascher 2004, see § 3.2.3.1).

Because such a large proportion of Victoria is private land, sympathetic whole-of-catchment for freshwater ecosystems will require effective partnerships between private landholders and all levels of local, regional and state institutions. Other enabling factors that might be required include legislative, institutional and administrative reform, statutory mechanisms and financial incentives. For instance, legislative and administrative reform is likely to be required to improve the management of riparian land and Crown Water Frontages (DSE 2009d). Various forms of financial incentives are being trialled to improve terrestrial and freshwater habitat and biodiversity values on private land (e.g. BushTender, RiverTender).

Whole-of-catchment management is necessarily a complex process, involving multiple actors across multiple jurisdictions, many of whom may have competing interests. Consistent alignment of decision-making at State, regional and local levels is a huge challenge. On this point, I note that Victoria has an array of high-level (i.e. State and regional) policies, frameworks and tools that are meant to guide whole-of-catchment management. They include:

a) Victorian River Health Strategy (DNRE 2002);
b) Securing Our Water Future Together: A Victorian Government White Paper (DSE 2004);
c) Securing Our Natural Future: A White Paper for Land and Biodiversity at a Time of Climate Change (DSE 2009d);
d) Regional Sustainable Water Strategies (DSE 2009d);
e) Regional Catchment Strategies; and
f) River Health Strategies

There is also a forthcoming Victorian Strategy for Healthy Rivers, Wetlands and Estuaries, due in 2011 (DSE 2009d). It is perhaps worth stepping back and asking how much duplication of effort there is and if such a profusion of policies, frameworks and strategies helps or hinders practical implementation of whole-of-catchment management.

4.2.1 Conjunctive Surface water-Groundwater Management

The Victorian Government has explicitly recognized the need for a landscape approach that not only encompasses the management of land under public and private tenure but considers the interplay between surface water and groundwater, and between freshwater and marine ecosystems (DSE 2009d). Under climate change, reliable surface water supply is likely to decrease and groundwater resource development is likely to intensify. However, increased groundwater withdrawals are only sustainable if it remains well below groundwater recharge and Kundzewicz & Döll (2009) caution that groundwater is not likely to ease freshwater stress in areas where climate change is projected to decrease groundwater recharge. Wise use will increasingly mean conjunctive surface water-groundwater management resources to ensure that use of groundwater does not adversely impact surface water resources, and vice versa (Palmer et al 2009).

Groundwater management is in a curious position. The *Water Act 1989* requires that the environmental water requirements be considered in determining the sustainable yields of groundwater systems. The major difficulty is that there is as yet, no accepted definition of GDEs. The water requirements of commonly recognized GDEs such as perennial streams and permanent wetlands in a floodplain system as well as riparian and floodplain forests and woodlands (see § 1) are not well understood and no consistent method yet exists for assessing their requirements. Consequently, there are no specific provisions for their protection or maintenance.

I note that information on groundwater resource availability and management in Victoria was dispersed and relatively difficult to access. The following material is largely based on the Australian Natural Resource Atlas (ANRA) which is an online publication of theme assessments undertaken by the National Land & Water Audit 2002 (http://www.anra.gov.au/topics/water/availability/vic/index.html).

A nationally agreed definition of sustainable yield is now available, but there is as yet no agreed methodology for determining sustainable yields. In Victoria, the current methodology adopted for estimating sustainable yield in Groundwater Management Areas (GMAs) involves checks on aquifer storage, river recharge/discharge, aquifer throughflow, well interference, seawater intrusions and pressure/head loss. The most commonly considered issues are baseflow to river systems and the intrusion of seawater. The sustainable yield methodology varies across the State according to the aquifer characteristics being investigated. In most cases, because of the lack of usage data and, in many cases bore hydrograph data, the sustainable yield has been determined as a percentage of rainfall, with adjustments made to take account of environmental requirements to the extent possible given currently available information. The methodology does not yet give explicit consideration to the water requirements of, and provisions for GDEs. It is claimed that efforts are made to ensure that sustainable yields are set to avoid significant interference with GDEs, but no details are provided.

It is however, acknowledged that the derived estimates of sustainable yield are relatively subjective. Until there is more substantial data on usage it will not be possible to derive water balances for the Groundwater Management Units (GMUs), and determine the recharge that provides the basis for sustainable yield. Similarly, the lack of information about the requirements of groundwater dependent ecosystems has meant that some fairly broad assumptions about these requirements have had to be made. It is claimed that because of these, and other uncertainties such as the impact of climate variability and the likely impacts of plantation forestry on sustainable yields, a conservative approach has been adopted in the estimation of sustainable yields for GMUs. However, there is no explanation of exactly what this conservative approach entails.

There are clearly some very serious knowledge gaps with regards to conjunctive surface water-groundwater management. The National Water Commission (NWC) has developed a National Groundwater Action Plan to address some of these issues. A $50 million National Groundwater Assessment Initiative has been set up to fund hydrogeological investigations Some of the research projects include:

- National assessment of surface water/groundwater connectivity
- National standards on groundwater mapping, definitions and assessment
- Atlas of groundwater-dependent ecosystems
- A consistent approach to groundwater recharge determination in data-poor areas
- Rollout and adoption of framework for assessing environmental water requirements of groundwater-dependent ecosystems

When it becomes available, it will be critical for the resulting information, standards and tools to be incorporated into the assessment of the sustainable yields of groundwater systems.

4.3 Connectivity

Heller & Zavaleta (2009) found in their review that the most frequently proposed recommendation for climate change adaptation was to improve connectivity, so that species can move. Workers variously recommended connectivity zones, corridor networks, 'biolinks' via riparian areas, railway reserves, shelterbelts, wildlife passages in infrastructure barriers and so on (see e.g. Opdam & Wascher 2004; DSE 2009d; Heino *et al*. 2009). Despite wide acknowledgement, these connectivity strategies were among the most poorly developed recommendations, limited mainly to common-sense reasoning and very general actions without little specific guidance on implementation (Heller & Zavaleta 2009). For instance, identification of kinds of actors that might need to be involved (e.g. reserve managers, policymakers, individuals) or information gaps. Furthermore, despite widespread favor for ecological networks, assessment of their effectiveness remains in its infancy (Heller & Zavaleta 2009).

As Hodgson *et al*. (2009) explain, quantifying connectivity and its effects is complicated and fraught with uncertainty. Firstly, connectivity can be defined from multiple perspectives. Functional connectivity is species specific and estimates the rate of (actual/potential) immigration into some point in the landscape (Hanski 1998). Being species-specific, functional connectivity obviously depends on a species' biology and ecology as well as its interaction with the distribution and quality of habitat in the landscape. This is both difficult and likely to entail a great deal of uncertainty. These challenges carry over to the task of quantifying the benefits of connectivity for biodiversity persistence (Hodgson *et al*. 2009). Structural connectivity generalizes the connectivity of habitat types without reference to any particular species. The assumption here is that structural connectivity is a reasonable proxy for functional connectivity of multiple species. However, Hodgson *et al*. (2009) contend that this assumption may become less tenable with climate change and induced changes to what currently constitutes 'habitat'. They argue that uncertainties in the estimation and effects of connectivity limit its value as a conservation metric and a more efficient approach is to target maintenance and improvement of habitat area and quality. This approach is likely to remain robust in the face of uncertainty from a range of sources (see § 4.1) and in any case, connectivity is often co-incidentally improved when habitat area and quality increase. In freshwater ecosystems, improving habitat area, quality and connectivity is of course, dependent on provision of adequate and appropriate flow regimes.

However, in situations where it is important to pursue this approach, this task which is likely to require some degree of water and land use reform will require the cooperation and coordination of multiple actors across multiple jurisdictions (e.g. State and local governments, catchment management authorities, rural water authorities, urban planners, community groups, conservation organizations and private landholders). Given the complexity of this challenge, detailed documentation and sharing of illustrative examples of current corridor projects or elaboration of specific ecological or political tactics for corridor creation will be hugely valuable for redressing the dearth of guidance on implementation. Accounts of failed initiatives, setbacks and surprises, all of which provide opportunities for learning are also equally valuable (Knight 2009). In the context of southeastern Australia, valuable examples we can learn from include the 'Alps to Atherton Initiative', 'Habitat 141', the River Murray-Coorong NatureLink and Project Hindmarsh.

4.4 Restoration & management

Experiences of restoration science throughout the world indicate that many of the profound changes that have been wrought in our riverine systems are extremely difficult to rectify (Pretty *et al*. 2003; Koenig 2009; Palmer & Filoso 2009) and examples of ecologically meaningful successes are rare (Palmer *et al*. 2005; Brooks & Lake 2007). In this section, I discuss two topics of critical importance for effective freshwater conservation, namely, environmental flows and riparian land and floodplain management.

4.4.1 Environmental Flows

Given the extent and profundity of impacts wrought by river regulation (Bunn & Arthington 2002; McMahon & Finlayson 2003), restoration of the natural flow regime, or at least some environmental flow provision and management is of paramount importance for effective freshwater conservation. This is clearly and unequivocally recognized by the National Water Commission's position statement on water-dependent ecosystems (NWC 2008). Commitments under the National Water Initiative (NWI) to water-dependent ecosystems, explicitly recognizes the difficult balance that must be struck between water for consumptive uses and water for the environment, in optimizing outcomes across environmental, social and economic values. Water planning is the fundamental means achieving this balance. At the most basic level, overallocated water systems need to be returned to environmentally sustainable levels of extraction and environmental water must be provided to water-dependent ecosystems to maintain freshwater biodiversity and ecosystem services. In this section, I concentrate on the issue of environmental water and discuss the issue of overallocation and environmentally sustainable levels of extraction in § 4.5.

The Victorian experience with Environmental Water Reserve (EWRs) is described in Box 2 and is instructive for understanding why the NWI calls for:

- Environmental water to enjoy the same security as water for consumptive uses
- Environmental water managers to be established and equipped with the necessary authority and resources
- Water market and trading arrangements to protect the needs of the environment
- Environmental water to be included in water accounts and audited
- Periodic assessments of river and wetland health to be conducted so that adaptive management can be undertaken on an evidence basis.

Box 2. Environmental Water Reserves (EWRs) in Victoria

The Environmental Water Reserve (EWR), introduced by the Victorian Government in 2005, provides legal recognition of the amount of water set aside to maintain the environmental values of water-dependent ecosystems. The EWR comprises three types of water: callable volumes in storage (entitlements), which can be released from storage by an environmental water manager to meet specific environmental needs; rules-based water such as passing flows; and rules-based, above-cap flows, which are released from storage, or made available to the environment by a storage operator or licensing authority (DSE 2009d). A mere 6% of the EWR consists of actual water entitlements in storage. The majority of the remaining 94% comes from passing flows, above cap water and reservoir spills (DSE 2009d). However, this water is highly vulnerable to the impacts of climate change (DSE 2009d). Environmental water management strategies that have been developed thus far include carrying over environmental water for use in future years, use of water at multiple sites through capture and reuse of returned flows and use of consumptive water en-route to maximize environmental outcomes (DSE 2009d).

The Victorian Government reports that significant volumes of environmental water have already been recovered and future water recovery projects are likely to substantially increase this volume. This includes water savings generated by infrastructure improvements such as 75 GL as part of the Northern Victorian Irrigation Renewal Project, 83 GL as part of the Wimmera Mallee Pipeline Project and 7 GL as part of the Macalister Irrigation District 2030 program (DSE 2009d).

What does all this mean in practice?

According to the 'State of the Environment Victoria 2008' report, "In many rivers and aquifers the current EWR is inadequate and vulnerable, placing environmental values at risk. Commitments to provide environmental water were qualified in 40 locations across Victoria in 2006-07, as part of drought contingency measures." (CES 2008). Furthermore, "In the Loddon, Murray, Campaspe and Goulburn Rivers, where the EWR has been recently boosted with low reliability water shares, low streamflows have meant this water is not yet available. This policy appears inconsistent with current projections for climate change, and appears to undermine a major benefit of creating the EWR, which was to give the environment an entitlement with legal status equivalent to that for water allocated to consumption. During times of low streamflow, the water allocation system reduces environmental flows more than it reduces water for consumptive uses." (CES 2008).

EWRs are a relatively new instrument and its design, governance structure and operational management is complex. Nevertheless, it is a critical resource for the protection, maintenance and conservation of water-dependent ecosystems. There must therefore be adequate provision, security of entitlement, clarity with respect to responsibilities, necessary authority and resources, accountability with respect to management and delivery and monitoring of outcomes within an adaptive management framework.

As the Victorian experience with EWRs (Box 2) shows, there is much room for improvement with regard to adequate provision and security of entitlement. For instance, given that major infrastructure investments have generated water savings and are expected to continue to recover substantial amounts of water for the environment, it should be possible to boost the percentage of (callable) actual water entitlements in storage and/or increase the reliability of entitlements. NWC (2008) noted that in spite of the legislation now passed in all jurisdictions, environmental water allocations often lack specificity and there is uncertainty around the status and security of environmental water holdings.

As for clarifying responsibilities, it appears that the present arrangement in which formal rights to the EWR are held by the Minster for the Environment, but operational management rests with the EWR officer of each Catchment Management Authority (CMA) is due to change. According to DSE (2009d), a new statutory body - the Victorian Environmental Water Holder will be created to manage environmental water across Victoria, and to coordinate watering programs with the Commonwealth Environmental Water Holder. It remains to be seen as to exactly how this move clarifies the responsibilities of CMAs. NWC (2008) noted that there are many shortcomings in the governance and operations of environmental water managers and that statutory empowerment, funding, skills and access to science, data and best practice guidelines all require urgent attention. The development of a national community of practice in environmental water management is an important initiative that will support these water managers.

With respect to monitoring and adaptive management, NWC (2008) reported that there is general deficiency in monitoring and reporting on plan implementation and this constitutes a significant weakness when coupled with gaps in ecological knowledge and the occurrence of climatic conditions outside the planned-for circumstances. They advocated more systematic monitoring and reporting to enable the water management regime to be adapted intelligently in the light of experience. Victoria is perhaps performing better on this front than other jurisdictions. The Victorian Environmental Flows Monitoring and Assessment Program (VEFMAP) is a large-scale, multi-basin program that was developed for the express purpose of monitoring the effects of environmental flow provision to eight high-priority river systems across Victoria (Chee *et al*. 2006; Webb *et al*., *in review*). This multi-agency program is now operational across all eight river systems with >100 sites being surveyed virtually simultaneously in spring, and other monitoring continuing year-round (Webb *et al*., *in review*).

The learning element is essential because the restoration of natural flow regimes in inherently variable freshwater ecosystems is complex and we do not yet have a good understanding of how partial delivery affects the ecological values we seek to protect and conserve.

4.4.2 Management of riparian land and floodplains

Riparian zones and floodplains are highly productive, ecologically important ecotones. Although riparian areas constitute a relatively small proportion of the total catchment area, they have a disproportionately large influence on the healthy functioning of river ecosystems (e.g. by providing habitat, shading and thermal refuges, contributing carbon and nutrient inputs) and require sensitive management. The significance and ecological functions and processes of floodplains have been described in § 1 and 2.1. Because of their functional importance numerous authors recommend targeting riparian and floodplain areas for restoration or rehabilitation (Palmer *et al*. 2009; Seavy *et al*. 2009).

On smaller streams in agricultural landscapes, riparian land is usually in private ownership. The State has a network of public riparian reservations known as Crown Water Frontages (CWFs), mostly on larger streams where the riparian land forms a boundary between properties (DSE 2009d). This network of reservations is unique to Victoria. CWFs have generally been licensed to an adjoining landholder for grazing purposes or for the cultivation of crops. More recently they have also been licensed for conservation purposes. There are currently about 10,000 licensed CWFs across Victoria. Licenses are issued for five year periods with renewals scheduled in 2009, 2014, 2019 and finally in 2024. Licensees are responsible for managing weeds, pests and fire on the frontage and for maintaining public access for recreation. Many CWFs are currently being used by adjacent landholders, without a license, for purposes that require licensing (DSE 2009d).

Much of the native vegetation across Victoria's riparian land has been cleared or degraded by grazing. This has led to a decline in the health of the riparian environment and in stream condition. Direct stock access to rivers and streams has further degraded the State's waterways and water supplies (CES 2008; DSE 2009d). The Victorian Government recognizes that efforts to improve the condition of riparian land need to be accelerated. Streams that include CWFs are often of the highest priority for waterway protection and restoration (DSE 2009d).

In the first instance, an effective measure to facilitate natural recovery where possible, is to reduce anthropogenic pressures in riparian and floodplain areas. This means minimizing land use impacts, for instance through cessation of grazing, fencing off riparian areas, low impact agriculture and weed removal and control. Riparian and floodplain protection and restoration efforts need to be accompanied by consistent, whole-of-landscape approaches within the entire contributing catchment area to give these actions the best possible chance to succeed. Unfortunately, a major opportunity to reform the management of CWFs was missed in the October 2009 renewal of CWF grazing licenses. Continued grazing along CWFs detracts from many State and regionally managed programs for sustainable land and water management such as those targeted at maintaining diverse flora and fauna, maintaining stream bank stability, improving water quality, regenerating native vegetation and weed control. The negative impacts of continued grazing in riparian land will continue to hamper and potentially undermine regional-level efforts at riparian, waterway and overall catchment management. This situation highlights the immense challenge of aligning decision-making at State, regional and local levels to achieve whole-of-catchment management.

4.5 Reduce threats & pressures

Direct pressures from water resource development and efforts to ensure security of supply in our highly-variable hydrological systems include dam construction, water extraction and flow regulation, stream channelization and desnagging, the draining of wetlands and construction of levees. Indirect pressures include native vegetation clearing in catchments, agricultural development and attendant effects of erosion, sedimentation, nutrient run-off and alien species introduction (CES 2008).

If we cannot prevent environmental change, we may be able to reduce the magnitude of its impact upon focal species. Reducing the impact may involve minimizing habitat loss, unseasonal flow patterns, pollution and fishing pressure (Kundzewicz *et al*. 2008; Palmer et al 2009). The point is to reduce the impact of any activity that might induce physiological burden or in some way of other affect basic survival, population processes and reproductive capacity. This really is a strategy for 'buying time' until effective solutions can be devised and implemented for underlying causes or to 'tide over' inevitable time lags following the implementation of solutions. As pointed out in § 4.1, the other (optimistic) view is that if the rate of decline can be slowed, populations have an enhanced "window of opportunity" in which to evolve adaptations to environmental stresses. This lengthens the time scale available for evolutionary change and provides more opportunity for evolution by natural selection to alter the species' niche sufficiently to ensure persistence in the novel environment (Holt & Gomulkiewicz 2004).

Any serious discussion of effective and durable solutions for addressing underlying causes must consider water planning, overallocation and environmentally sustainable levels of extraction. The National Water Commission's 2007 First Biennial Assessment of Progress in the Implementation of the National Water Initiative (NWI) found that all states had made statutory provision for water to meet environmental and public benefit outcomes within water plans, however:

- overallocated systems were not always adequately identified
- environmentally sustainable levels of extraction were poorly defined
- there was considerable variability in the quality of the science underpinning water plans
- in many cases the trade-offs between environmental and consumptive uses were not transparent
- there was often a lack of specificity in the environmental outcomes

The NWC has an ambitious and well-targeted program of research activities for addressing knowledge gaps. However, improved knowledge alone is a weak instrument for change (Jacquet 2009) and the NWC recognizes that the recovery of overallocated systems, continues to be a major challenge in implementing the NWI Agreement (NWC 2008). To this end, NWC (2008) has adopted two major priorities to:

a) *Help develop and implement national guidelines and procedures for determining environmentally sustainable levels of extraction of water.*
A nationally agreed method will expedite the formulation of water plans that protect water-dependent ecosystems and include a pathway to recover overallocated systems. The methods will include guidelines for establishing clear environmental outcomes.

b) *Pursue an agreed national inventory of over-allocated water systems together with commitments by governments to return them to sustainable levels of extraction.*

> Identifying overallocated systems and recording agreed actions to recover the water needed to restore sustainability is central to achieving environmental outcomes contained in the NWI.

Given the expected time lags between mitigation or remediation and ecosystem response, dynamic threats must be monitored in the intervening period to allow timely responses for safeguarding investments where necessary. Climate change is expected to produce profound, cascading effects in ecological, social and economic systems and their interactions. The nature and trajectories of these changes is poorly understood and highly uncertain. To date, the implications of these dynamic threats for freshwater biodiversity have received little attention. Despite the inherent uncertainties, it is imperative to develop strategic capabilities for anticipating, monitoring and responding to dynamic and emergent threats. Practical tools should include mapping patterns and monitoring rates of spread of threats to biodiversity, as it is such threats to which conservation planning should respond (Margules & Pressey 2000). A better understanding of the present and future distribution patterns of various threats will help focus limited conservation resources on areas and features most at risk. It will also clarify the extent to which conservation priorities overlap with priority areas for extractive and destructive uses (Margules & Pressey 2000). Tools such as ecological risk assessment, species distribution models, hydrological and landscape dynamics models can be used to address this task.

4.6 Refugia

This section distinguishes between refugia at short- and longer-term timescales. In the short term, refugia from natural disturbances such as floods and droughts are important for ensuring population persistence, even if they are only used occasionally (Lake 2000). In effect, they confer spatial and temporal resistance and/or resilience to populations (Magoulick & Kobza 2003). Many forms of refugia (e.g. thermal, hydraulic/velocity and drought) have been lost through anthropogenic interventions such as the removal of streamside vegetation, channelization, desnagging and altered natural flow regimes. Numerous workers argue that restoring such refugia is important for facilitating resilience to ongoing anthropogenic disturbances (Lake 2000; Bond & Lake 2005; Lake *et al*. 2007; CES 2008; Ormerod 2009).

From the perspective of a longer timescale, two different types of refugia are important to consider - *stationary* refugia (also known as evolutionary refugia), where species are able to survive in regions that escaped the more dramatic climatic extremes; and *displaced* refugia, where species might find suitable habitats after they had been displaced by climate changes from their original location (Araújo 2009). The wholly subterranean aquifer and cave groundwater ecosystems mentioned in § 1 are an example of stationary refugia for stygofauna. I am however, not aware of examples pertaining to surface water ecosystems. The rationale behind stationary or evolutionary refugia is that refugia that protected species during climate shifts in the past are anticipated to be important sources for species re-colonization and radiation in the future (see references in Heller & Zavaleta 2009; Klein *et al*. 2009). Displaced refugia may occur in mountain ranges, deep valleys and areas with steep climatic and environmental gradients that are able to maintain pockets of environmental and climatic conditions that become regionally restricted (see also § 4.1).

With respect to Victoria, Dunlop & Brown (2008) claim in their report on 'Implications of Climate Change for Australia's National Reserve System: A Preliminary Assessment' that "Many authors also suggest areas acting as refuges from past climate change, climatic variation and disturbance should be reserved as a priority. Brereton *et al*. (1995) found many such areas are already included in the reserve system in Victoria." This is not an accurate statement of the work of Brereton *et al*. (1995). Brereton et al. (1995) found that "potential greenhouse refugia show a reasonable correlation with the current reserve system. However, in the west and central areas these are presently islands within vastly modified environments. The usefulness of these refugia is dependent upon long term access. A series of facultative corridors, or biolinks is needed." Care is required for interpretation here. The Brereton et al. (1995) study examined the potential effect of enhanced greenhouse climate change on the distribution of 42 species of fauna, selected from the major Victorian bioclimatic regions and ecosystems and from species considered most at risk from enhanced greenhouse climate change. The spatial scale of analysis was coarse, being based on blocks/grids of 6 min of latitude and longitude (6′ X 6′) (where latitudes and longitudes represented the block's centre and elevation was the mean for the block). The area of a 6′ X 6′ block varies with latitude but in central Victoria (37″S) is approximately 100 km^2. The climate scenario used included: +3 ^{0}C; +10% precipitation in summer and -10% precipitation in winter (*cf* more recent projections presented in § 2.2.1). Blocks which supported the bioclimates of 9 or more species under the scenario, were identified and tentatively assigned as potential 'greenhouse refugia'. This does not constitute an appropriate basis for asserting that climate change refuges are already included in the reserve system in Victoria.

4.7 Translocation & Reintroduction

Even with good landscape and hydrologic connectivity, some species for one reason or other will not be able to migrate to seek more favourable environment conditions. For instance, if they are dispersal limited, restricted to rare habitat types, have no pathways for moving to an adjacent catchment or if the rate of environmental change is too rapid. For such species, translocations from within their current range to locations suitable in the future are widely advocated (see references in Heller & Zavaleta 2009).

Translocations are however, a contentious issue because of the challenges associated with moving populations successfully and predicting suitable future habitats, as well as the potential for unintended consequences from introducing new species into existing communities (McLachlan *et al*. 2007). Empirical evidence suggests that animal translocations tend to be unsuccessful and costly (Fischer & Lindenmayer 2000). Proper evaluations of the feasibility of translocations requires stronger understanding of best available methods, potential risks, and policies for regional coordination to avoid situations in which different conservation objectives are put in conflict (McLachlan *et al*. 2007).

Nevertheless, translocation and reintroduction can work and have their place within the suite of options for conservation and adaptation. Two examples in Victoria, relating to the freshwater catfish and the barred galaxias illustrate this. As mentioned in a supplementary note in Table 2, the translocation of the *FFG*-listed freshwater catfish to the Wimmera River catchment in the 1970s has culminated in the establishment of a self-sustaining population that now constitutes a stronghold in a region where it was not endemic. The nationally endangered barred galaxias is known from only 12 small populations in a small area of Victoria that was damaged in the severe February 2009 bushfires. Loss of streamside vegetation and cover and heavy rains following bushfires can cause an influx of ash and soil into rivers and streams with detrimental impacts on water quality and food supply for the barred galaxias. 394 individuals were rescued from their habitat and have been moved to Department of Primary Industry and Department of Sustainability and Environment facilities until conditions in the wild become suitable for reintroduction.

5 Conclusion

Quantitative methods and tools are important for decision support, but are ultimately of limited efficacy without equivalent investment of intellectual capital and effort in bridging the 'knowing-doing' gap between research and implementation. As Knight *et al*. (2009) emphasize, "Spatial prioritization techniques simply provide tools that help people to articulate their goals and to make decisions; alone, they do not deliver conservation action and so they must be complemented with social, political and institutional tools and processes."

The multiple use nature of freshwater resources and ecosystems firmly embeds them within our complex social-ecological systems. How might we construct and foster the societal infrastructure necessary to develop coherent, coordinated, wise-use policies with broad societal support? In his analysis of 'Civil Society and Resource Use', Rogers (2006) observed that, "In the modern day, many components of civil society have become divorced from the understanding that they are dealing with common property resources. … legislation leads to systems of land tenure and access to resources dominated by private ownership. A particular effort is needed to regenerate this understanding and much of it needs to come from civil society itself. Some important mind-set changes are needed to give a renewed sense of cooperation in the use of natural resources that provide common property goods and services. The ultimate responsibility for this mind-set change probably falls on a very wide range of formal and informal civil institutions (CMAs, opinion makers, the media, education institutions etc.) but it is scientists and service agencies that have the proximate responsibility for deepening the thinking within these institutions."

In addition, Rogers (2006) highlights two particular points which need to be broadly appreciated by all parties:

1. Society on the whole have very little appreciation for the problems managers experience as a consequence of variability in the goods and services that the resource base delivers. But notwithstanding this, society has very high expectations of managers. Compounding the difficulty is that "managers have a very limited toolbox with which to change resource use patterns and so achieve desired distributions of the costs and benefits that accrue." Most of these tools are technological hardware for 'fixes' that are generally better suited to treatment of symptoms rather than causes (e.g. equipment for riparian replanting, fish ladders).

2. Modern society has largely transferred the responsibility for solving environmental problems to public agencies and in doing so, has implicitly absolved itself of the risk of failure and transferred it to the public agencies where, in turn, it is often carried by individuals whose jobs are on the line. Rogers (2006) comments that "neither scientists nor civil society seem to really appreciate how difficult it is for public agencies to balance the power they need to take action in solving problems, with the humility they need to act in the service of society. All parties need to understand that partnerships are needed to produce broad societal response to environmental problems and that such a response requires a better distribution of power, risks, humility and rewards across participants."

He concedes that getting these two points across will be an exceptionally difficult task for scientists and both service and civil institutions but that the key lies in the processes used to develop a common understanding and collective decision-making in the redistribution of costs and benefits of resource use. An important step in this process is to prepare government management institutions and society "to engage the knowledge, the problems and the solutions needed to achieve some collectively defined set of future conditions" (Rogers 2006). And to engage all stakeholders in problem-solving situations as co-learners to facilitate partnership building and foster the relationships required to develop coherent, coordinated, wise-use policies with broad societal support. This was something that had to be directly grappled with in the collaborative implementation process of the Victorian Environmental Flows Monitoring and Assessment Program (VEFMAP) (see § 4.4.2). This experience has been documented and we hope that this account of the complexities and lessons from our real-world example will both encourage and ease the process for others attempting a related undertaking (Webb *et al.*, *in review*).

Because there are no stopping rules in solving complex ('wicked') environmental problems, continual learning amongst all parties is imperative if decision-making is to be responsive and adaptive (Webb *et al.*, *in review*). We also require what Rogers (2006) calls "self-reinforcing processes that maintain synergy, focus and momentum in a partnership." Learning frameworks are one way to support this. They include social learning (Parson & Clark 1995; Knight *et al*. 2006; Rogers 2006; Smith *et al*. 2009) and learning communities or communities of practice (Rogers 2006; NWC 2008). Social learning aims to facilitate problem ownership, value formation, discussion, deliberation, risk assessment, negotiation and reconciliation of interests (Lee 1993; Sagoff 1998). Its is characterized by people modifying their behaviour in response to lessons learned when undertaking thoughtful and often collective action and is aimed at increasing human capacity to solve problems and adapt to changing conditions (Rogers 2006; Knight *et al*. 2006). In learning communities or communities of practice, similar principles apply and individuals accept responsibility for collective learning of the group above their own vested interests and foster a cooperative, reflective and experimental approach to learning (Rogers 2006).

Decision-making in ecosystem management is a process of balancing multiple objectives, constraints, trade-offs and uncertainties against a complex backdrop of socio-economic, cultural and political considerations and limited ecological understanding. Developing effective, durable solutions requires cooperative decision-making founded on broad societal participation.

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THE UNIVERSITY OF MELBOURNE

ENVIRONMENTAL STREAMS DATABASE OF VICTORIA

METADATA REPORT

VERSION 1.0
DECEMBER 2009

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THE UNIVERSITY OF MELBOURNE

Environmental Streams Database of Victoria

PREPARED BY YUNG EN CHEE

SUMMARY

This document describes the development and contents of a fine-scale, stream link-based GIS database for Victoria. The database comprises an extensive set of bioclimatic, physiographic, edaphic, land cover and disturbance-related attributes at multiple spatial scales. The principal motivation for developing this GIS stream network database was to enable characterization, representation and modeling analysis of the multidimensional environmental space of freshwater-dependant biota.

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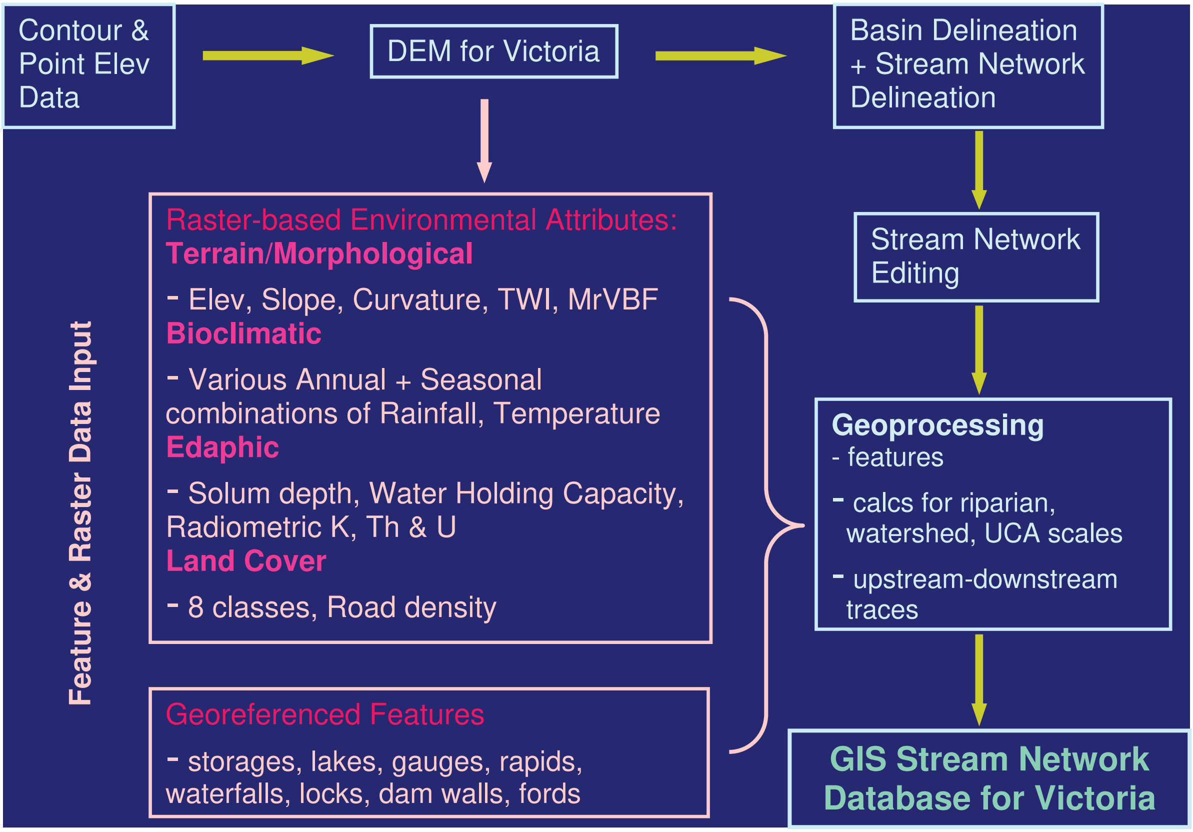
The backbone of the stream network database, is an ordered, link-node representation of the stream network for Victoria. A simplified overview of the data inputs and iterative steps and processes involved in the development of the ESDV is shown in Figure 1. I tried to incorporate as many ecologically relevant attributes as possible from a range of physiographic, bioclimatic, edaphic, land cover and human disturbance-related variables (see Table 1). Note that it's possible to incorporate virtually any geospatial environmental data. The current suite of available attributes can be extended with additional geoprocessing. The source or derivation of key data input sources for the construction of the ESDV are summarized in the Appendix.

Figure 1. Simplified overview of data inputs and iterative processes used to develop the Environmental Streams Database for Victoria. Refer to text for methodological details.



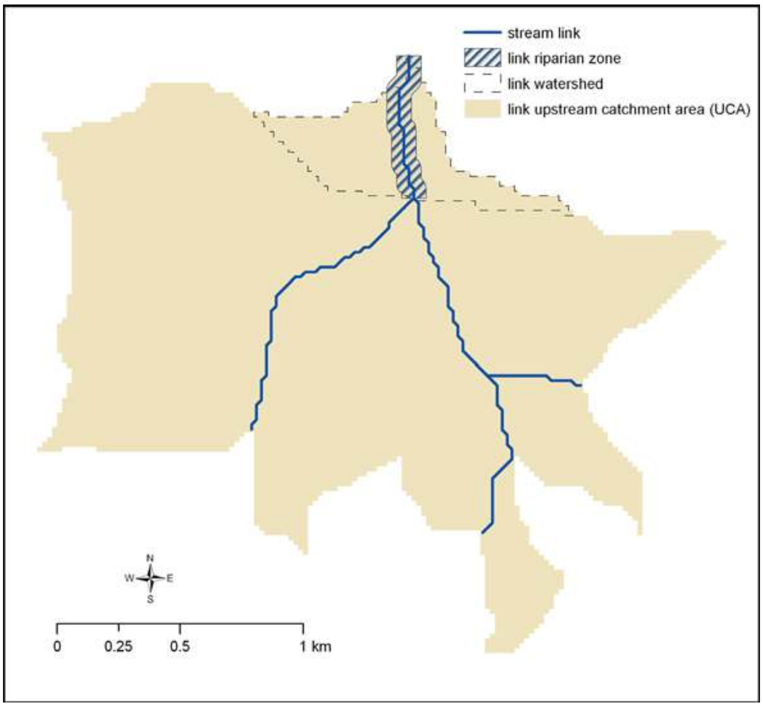
A hydrologically-correct 20 m digital elevation model (DEM) (see Appendix) was constructed from contour and point elevation data sourced predominantly at 1:25,000 map scale (VicMap Elevation, Corporate Geospatial Data Library, DSE). Using the DEM, the stream network was automatically delineated using ArcGIS (ESRI Corporation, Redlands, California, USA) hydrological processing routines and possesses topologically consistent characteristics such as directionality and connectivity.

The stream link (i.e. the river segment between any two stream junctions) constitutes the basic ‘unit’ in the network and the minimum area used to define a stream link was 0.004 km^2. Watersheds for each stream link were also automatically delineated used the ArcGIS hydrological processing routines.

In flat, low-lying areas with little topographic variability, automatic drainage analysis of the DEM resulted in two significant problems: a) it produced output drainage that was unrealistically dense and b) it was not able to represent anabranching patterns. To produce reasonable channel networks in such areas, channel alignment had to be manually specified. This highly labour-intensive stream editing process was carried out using editing tools in ArcMap, guided mainly by stream line features digitized from 1:25,000 scale topographic maps (VicMap Hydro, Corporate Geospatial Data Library, DSE) and some terrain analysis raster datasets.

To characterize different aspects of the complex environmental space arising from longitudinal and lateral mosaic influences on riverine ecosystems, a comprehensive set of estimates of physiographic, bioclimatic, edaphic, land cover and anthropogenic disturbance-related variables, which were considered to have ecological relevance, were computed for every stream link at one or more, hierarchically-nested spatial scales (Table 1). The three scales were: a) the riparian zone with a width of 50 m on either side of a link; b) the immediate watershed of a link and c) the entire upstream contributing area associated with a link (Figure 2).

Figure 2. Example showing nested spatial scales at which environmental variables for individual links were computed. Flow direction is to the north.



Data inputs for the environmental variables were derived from a range of sources. Physiographic variables were calculated directly from the DEM or from rasters created by implementing terrain analysis algorithms for computing secondary terrain attributes such as topographic wetness index (Moore *et al*. 1993) and multi-resolution valley bottom flatness index (Gallant & Dowling 2003).

Bioclimatic variables were derived from interpolated surfaces estimated using the software package ANUCLIM 5.1 (Houlder *et al*. 2000) which uses thin plate smoothing splines fitted to long-term meteorological station data (see Figures 3 & 4 for examples).

Edaphic variables were derived from rasters of modeled solum depth (see Figure 5 for example) and plant available water holding capacity extracted from the Soil Hydrological Properties of Australia spatial dataset prepared by Western & McKenzie (2006). Radiometric data measures natural gamma radiation emanating from the earth's surface to a depth of about 30 cm. This gamma radiation is split into four channels - total radioelement count and percentage/concentrations of three naturally occurring elements: potassium (K), thorium (Th) and uranium (U). The varying concentrations and distribution of radiometric K, Th and U provide an indication of soil and rock characteristics and such data are used to assist with geological and soils mapping, mineral and petroleum exploration and land management. Rasters of radiometric K, Th and U data were obtained from datasets compiled from data acquired by the Geological Survey of Victoria, Geoscience Australia and private companies (Corporate Geospatial Data Library, DPI) (see Figures 6 and 7 for examples).

Land cover variables were derived from eight land cover categories obtained from the Modelled Native Vegetation Extent spatial dataset (NVE2007, Corporate Geospatial Data Library, DSE)(see Figure 8). For the disturbance-related variables, an input raster of road density was created using ArcGIS tools and road features captured at a 1:25,000 scale (VicMap Transport, Corporate Geospatial Data Library, DSE).

Figure 3. Raster surface of mean annual precipitation computed using ANUCLIM 5.1.

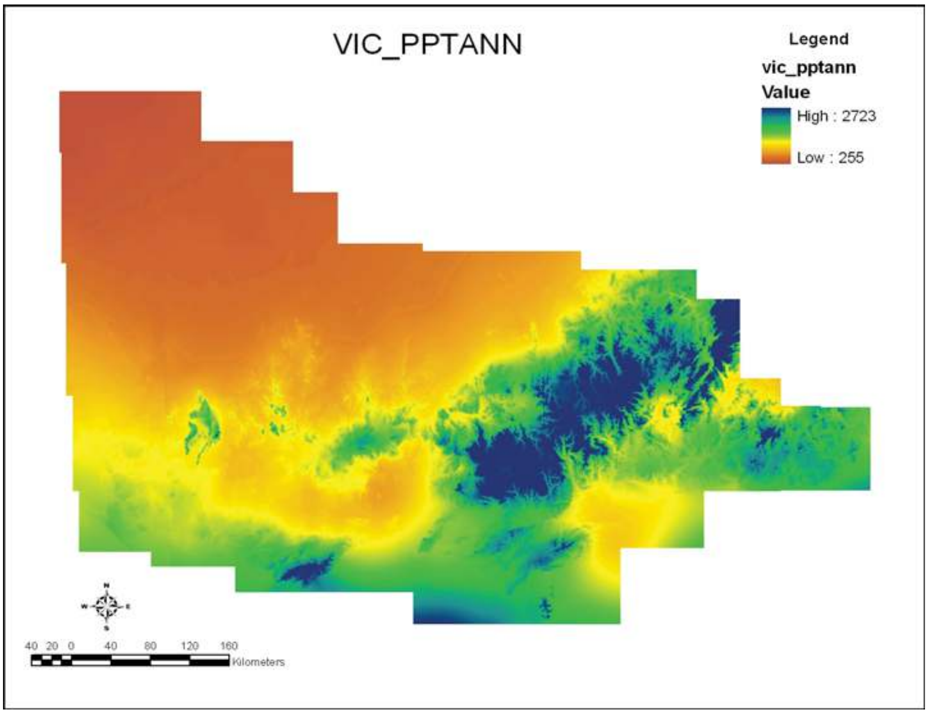


Figure 4. Raster surface of mean annual temperature computed using ANUCLIM 5.1.

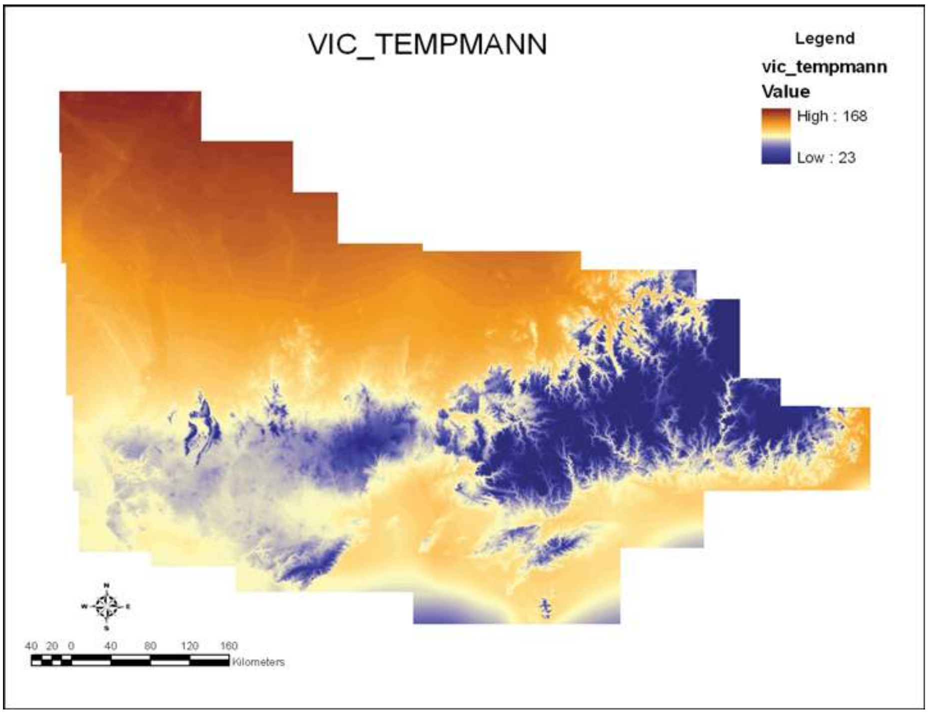


Figure 5. Raster surface of modeled solum depth (Western & McKenzie 2006).

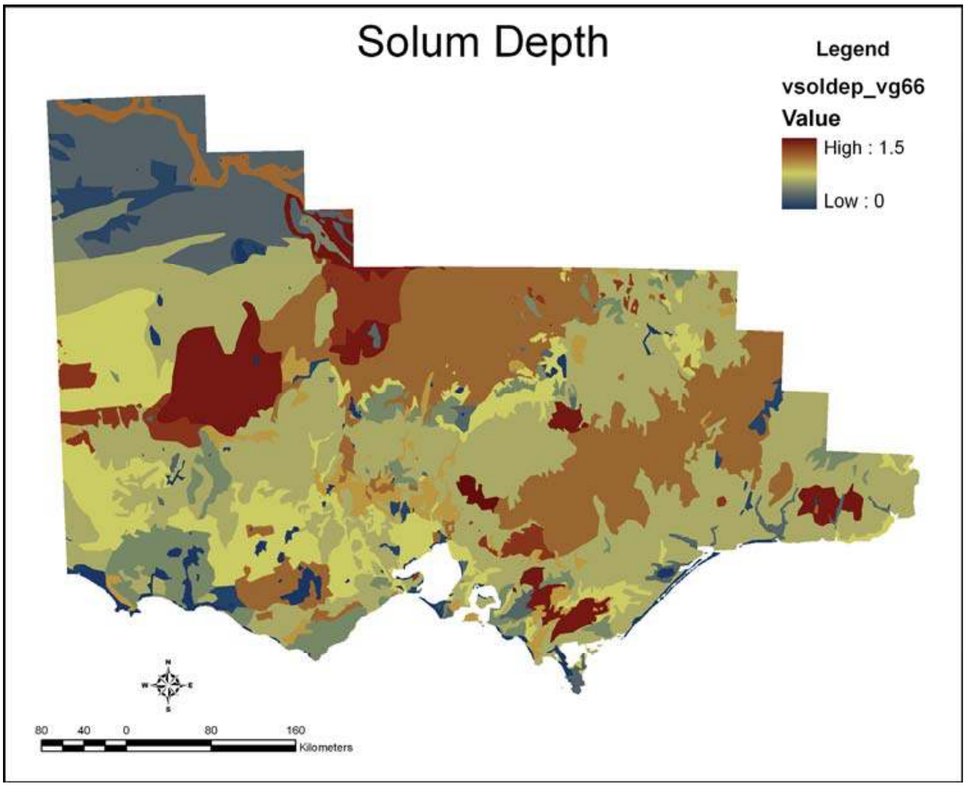


Figure 6. Raster of radiometric potassium (K).

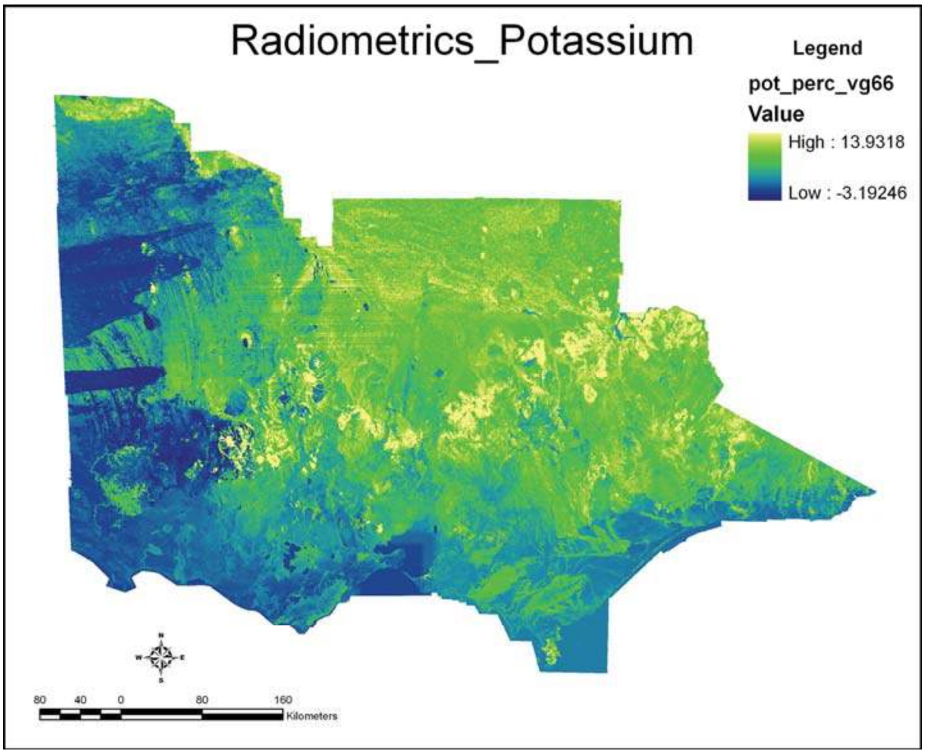


Figure 7. Raster of radiometric uranium (U).

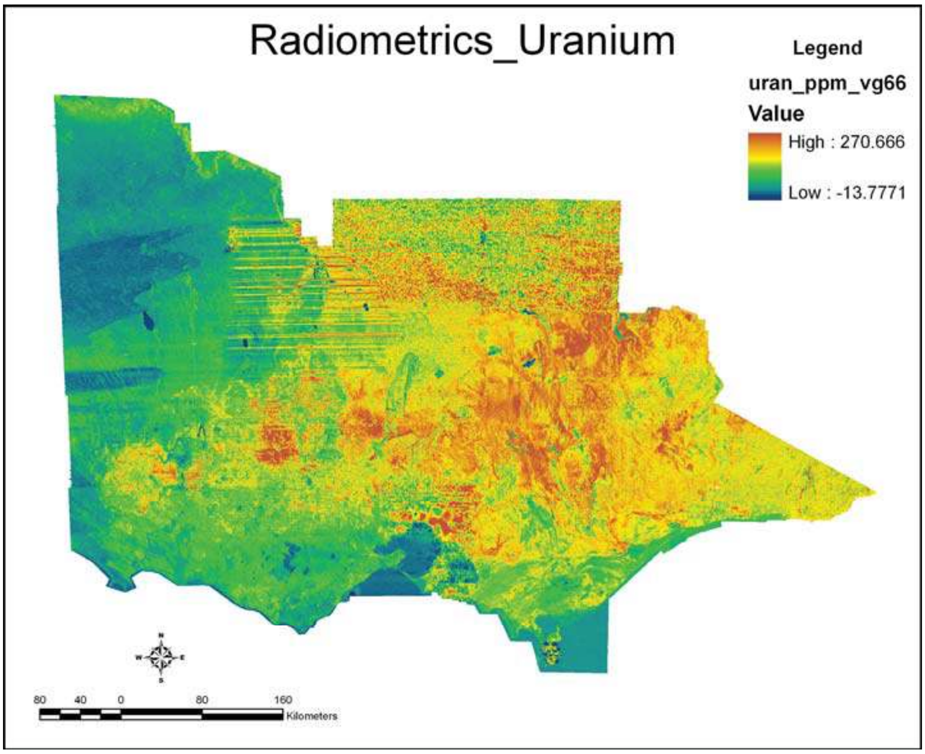
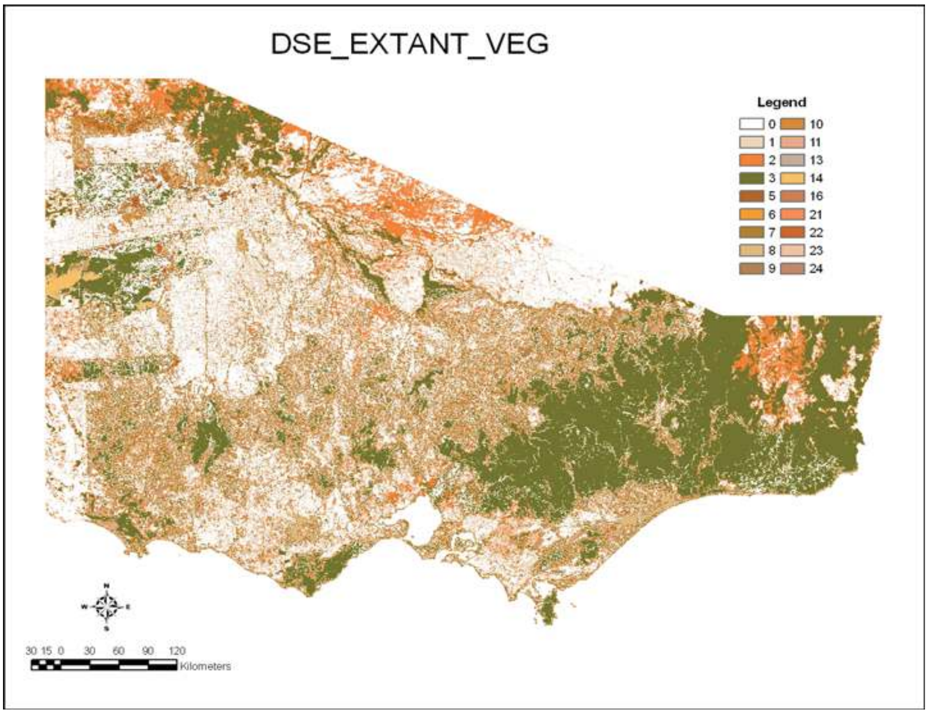


Figure 8. Raster of Modeled Native Vegetation Extent spatial dataset (NVE2007, Corporate Geospatial Data Library, DSE) used to define land cover classes.



Georeferenced data representing instream structures and features of potential influence on fish, such as rapids, waterfalls, dam walls, gauging stations and fords were obtained from spatial datasets VicMap Hydro and Transport (Corporate

Geospatial Data Library, DSE) and Thiess Services Pty Ltd (managers of Victoria's network of stream gauging stations). See Appendix for additional details. These features were spatially joined to the stream network so that they could be explicitly flagged in each affected link and used in computing other variables (see below and Figure 9).

Variable estimates for each link were computed at the riparian, watershed and upstream contributing area (UCA) scales using a suite of custom scripts written in ArcINFO AML. In essence, variables at watershed-scale were quantified by overlaying watershed boundaries on the various environmental raster datasets. Riparian-scale variables were computed in a similar manner using buffer boundaries generated along each stream link. UCA-scale variables were computed from watershed- or riparian-scale variables using network accumulation algorithms. The general modeling approach for these operations followed that of Wilkinson *et al*. (2004). Finally, the ordered, link-node representation of the stream network was exploited to conduct 'traces' in upstream-downstream directions along the flow path of each link, computing a range of variables of potential ecological relevance using purpose-written tracing AMLs. For example, estimates of the maximum slope encountered along a link's upstream flow path or the mean riparian tree cover along a link's downstream flow path (US_MAXSLOPE and DS_AVGRIPTRECOV respectively in Table 1).

The result of all these geoprocessing operations was the fluvial equivalent of "terrestrial focal predictors that summarize information on the neighbouring landscape within the focal cell" (Guisan & Thuiller 2005). Figures 10 and 11 show the range of environmental attributes that have been computed for a single selected stream link. The environmental streams database for Victoria consists of a total of about 400,000 links and about 100 environmental predictor variables. The environmental predictor variables are described in a little more detail in Table 1.

Figure 9. An excerpt of a stream network with its associated watersheds. Red circles denote flow gauging stations and pale blue circles indicate dam walls.

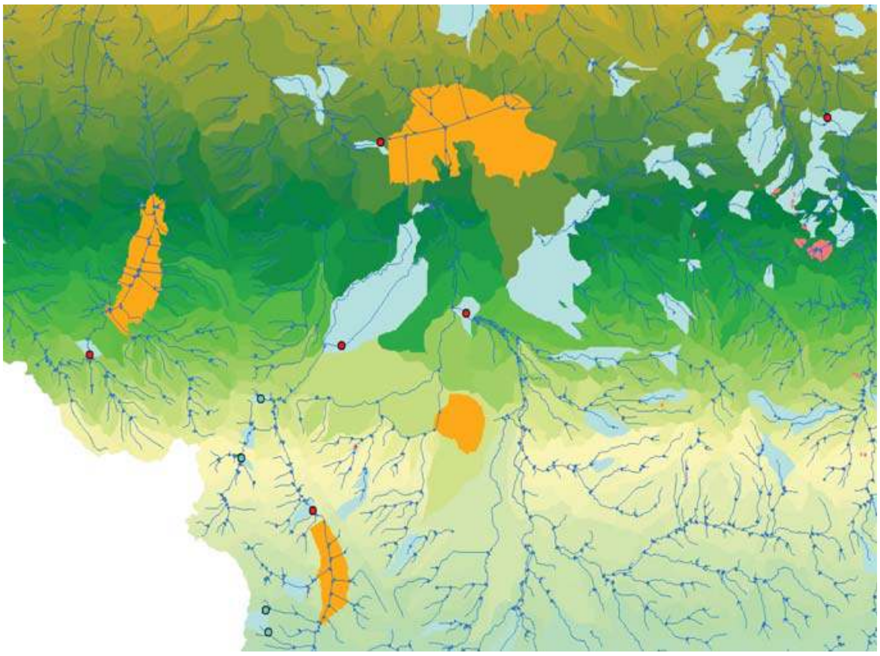


Figure 10. Range of environmental attributes that have been computed for a single selected (highlighted) stream link. Attributes shown in the left-hand panel are mostly terrain and bioclimatic variables.

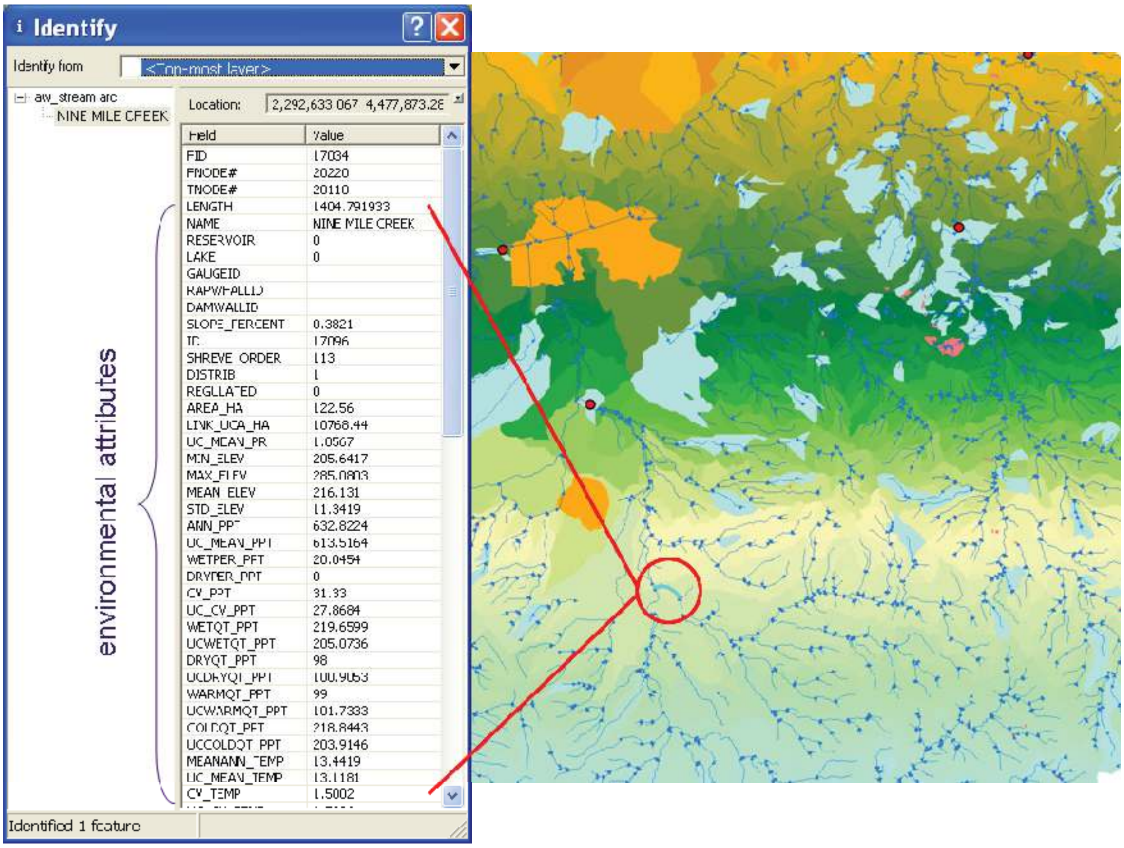


Figure 11. Range of environmental attributes that have been computed for a single selected (highlighted) stream link. Attributes shown in the left-hand panel relate mostly to land cover variables. The environmental streams database for Victoria consists of a total of about 400,000 links and about 100 environmental predictor variables.

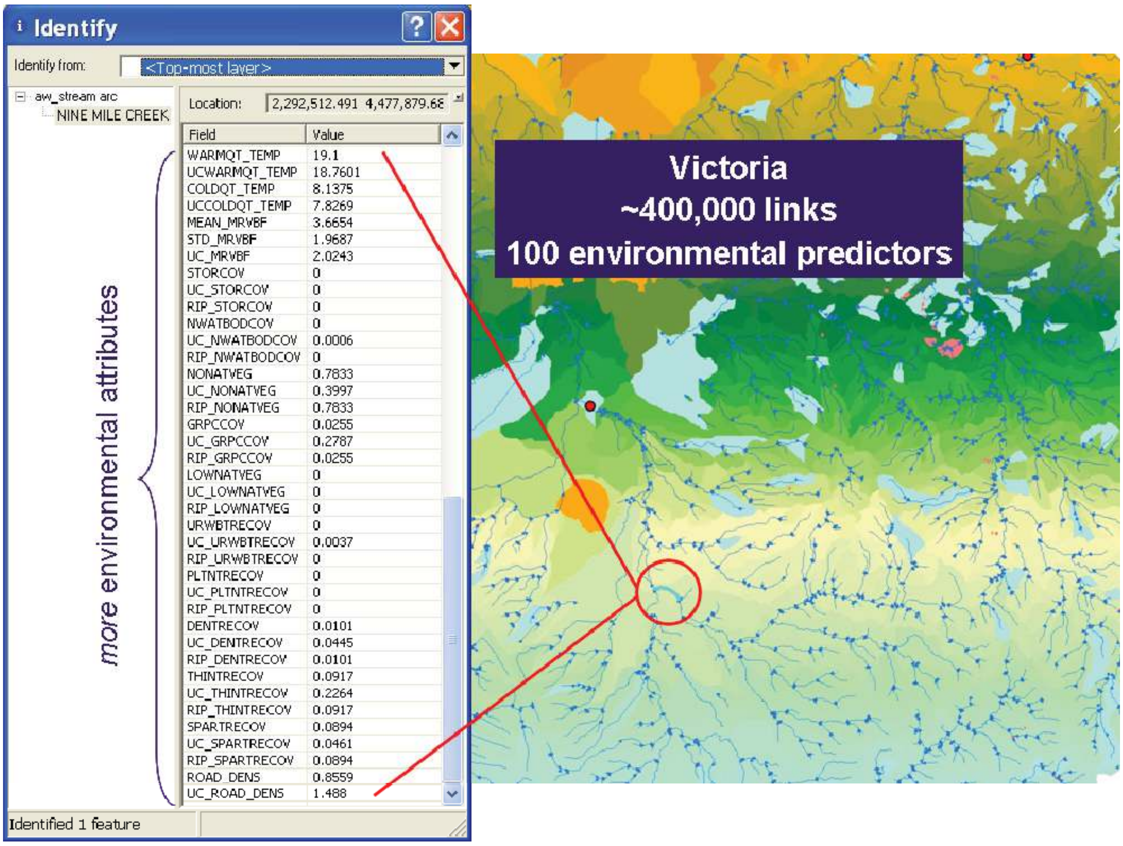


Table 1 provides an inventory of the environmental attributes available in the Environmental Streams Database for Victoria.

Table 1. Inventory of physiographic, bioclimatic, edaphic, land cover and human disturbance-related variables in the Environmental Streams Database of Victoria. The rules of nomenclature were devised to indicate the particular spatial scale (see Fig. 2) at which an environmental attribute for a link was computed. They are as follows: a) attributes with a UC prefix or UCA suffix indicate that the attribute relates to a link's upstream contributing catchment area (UCA); b) attributes with a RIP prefix indicate that the attribute relates to the riparian zone within the link watershed; c) attributes with US or DS prefix indicate that the attribute relates to a trace along the link's upstream or downstream flow path, respectively.

Item#	Item_Name	Variable Description	Comment
1	FNODE#	'From' node of link	
2	TNODE#	'To' node of link	
3	LPOLY#	utility	
4	RPOLY#	utility	
5	<STRM>#	utility	
6	<STRM>-ID	utility	
7	NAME	name of watercourse	**Inaccuracies present. Defer to names in the 1:25,000

			'HY_WATERCOURSE_polyline' shapefile in VicMap Hydro, Corporate Geospatial sData Library, DSE.
8	LINK_IN_OUT	utility	
9	DISTRIB	values < 1 denotes an anabranch link (specifies distribution value for certain attributes)	
10	RESERVOIR	1 denotes link within a reservoir/storage	
11	LAKE	1 denotes link within lake/waterbody	
12	GAUGEID	stream gauging station identifier (if any)	
13	GAUGE_OCC	1 denotes presence of gauge in link	utility –used to compute number of gauges along link upstream/downstream flow paths
14	RAPWFALLID	rapid or waterfall identifier (if any)	
15	RWF_OCC	1 denotes presence of rapid/waterfall in link	see entry for GAUGE_OCC
16	LOCKID	lock identifier (if any)	
17	LOCK_OCC	1 denotes presence of lock in link	see entry for GAUGE_OCC
18	DAMWALLID	dam wall identifier (if any)	
19	DMW_OCC	1 denotes presence of dam wall in link	see entry for GAUGE_OCC
20	FORDID	dam wall identifier (if any)	
21	FORD_OCC	1 denotes presence of ford in link	see entry for GAUGE_OCC
22	REGULATED	1 denotes regulated flow link	
23	ID	unique link ID (within the REGION)	
24	LENGTH	link length in metres	
25	SLOPE_PERCENT	link slope_percent value. Calculated as rise/run * 100	
26	SHREVE_ORDER	link shreve order	*see section on Data Limitations, Accuracy & Completeness
27	STRAHLER_ORD	link Strahler order	*see section on Data Limitations, Accuracy & Completeness
28	MIN_ELEV	minimum elevation in link watershed (m)	
29	MAX_ELEV	maximum elevation in link watershed (m)	
30	MEAN_ELEV	mean elevation in link watershed (m)	
31	STD_ELEV	standard deviation of elevation in link watershed	
32	AREA_HA	link watershed area (ha)	
33	LINK_UCA_HA	link upstream contributing catchment area (UCA) (ha)	
34	MIN_ELEV_UCA	minimum elevation in link UCA (m)	
35	MAX_ELEV_UCA	maximum elevation in link UCA (m)	
36	MEAN_ELEV_UCA	mean elevation in link UCA (m)	
37	HYPSOMETRIC	approx hypsometric integral – relative relief ratio of mean elevation of link UCA to total elevation range of link UCA. In theory, hypsometric integral values range from 0 to 1. Strahler (1952) interpreted it as a measure of the erosional state or geomorphic age of the catchment with low values representing old, eroded landscapes and high values as younger, less eroded landscapes. Dimensionless.	

38	TOTLENGTH_UCA	sum of length of all links within link UCA (km). Required for drainage density (see below) calculation	
39	LENGTH2_UCABND	maximum channel length to link UCA boundary (km)	
40	DRAINAGE_DENS	drainage density in link UCA (km/km^2)	see section on Data Limitations, Accuracy & Completeness
41	MEAN_MRVBF	mean multi-resolution valley bottom flatness (MrVBF) index value in link watershed. MrBVF index classifies degrees of valley bottom flatness based on integrating estimates of 'flatness' and 'lowness' computed at a range of scales. MrVBF is an expression of local relief in terms of valley confinement and floodplain extent with values typically ranging from 2.5 in narrow, confined valleys to ≥ 8 in broad floodplains. Threshold values of 4-4.5 are often used to designate floodplains. See Gallant & Dowling (2003) for full details.	Computed using MrVBF rasters derived using code provided by John Gallant. (Version current in Aug 2007). Ref: Gallant & Dowling (2003)
42	STD_MRVBF	standard deviation of MrVBF index value in link watershed	see entry for MEAN_MRVBF
43	UC_MRVBF	mean MrVBF index value in link UCA	see entry for MEAN_MRVBF
44	FLOODWIDTH_M	link floodplain width (m)	see entry for MEAN_MRVBF Computed in conjunction with MrVBF raster. For stream links of shreve order ≤3, areas within 200m of the network with MrVBF values ≥ 4.5 were considered to be floodplain. For stream links of shreve order >3, areas within (up to) 3km of the network with MrVBF values ≥ 4.5 were considered to be floodplain.
45	MEAN_TWI	mean TWI (topographic wetness index) value in link watershed	Computed from TWI rasters derived using ArcInfo scripts
46	UC_TWI	mean TWI (topographic wetness index) value in link UCA	as above
47	ANN_PPT	mean total annual ppt in link watershed (mm/yr)	Derived from bioclimatic rasters computed using ESOCLIM in the software package ANUCLIM with a 20m DEM as input data
48	WETPER_PPT	mean ppt of wettest week in link watershed (mm)	see entry for ANN_PPT
49	DRYPER_PPT	mean ppt of driest week in link watershed	see entry for ANN_PPT
50	CV_PPT	mean coefficient of variation of mean annual ppt in link watershed	see entry for ANN_PPT
51	WETQT_PPT	mean ppt of wettest quarter (any 13 consecutive weeks) in link watershed	see entry for ANN_PPT
52	DRYQT_PPT	mean ppt of driest quarter (any 13 consecutive weeks) in link watershed	see entry for ANN_PPT
53	WARMQT_PPT	mean ppt of warmest quarter (any 13 consecutive weeks) in link watershed	see entry for ANN_PPT
54	COLDQT_PPT	mean ppt of coldest quarter (any 13 consecutive weeks) in link watershed	see entry for ANN_PPT
55	UC_MEAN_PPT	mean total annual ppt in link UCA	see entry for ANN_PPT

		(mm/yr)	
56	UCWETQT_PPT	mean ppt of wettest quarter (any 13 consecutive weeks) in link UCA	see entry for ANN_PPT
57	UCDRYQT_PPT	mean ppt of driest quarter (any 13 consecutive weeks) in link UCA	see entry for ANN_PPT
58	UCWARMQT_PPT	mean ppt of warmest quarter (any 13 consecutive weeks) in link UCA	see entry for ANN_PPT
59	UCCOLDQT_PPT	mean ppt of coldest quarter (any 13 consecutive weeks) in link UCA	see entry for ANN_PPT
60	MEANANN_TEMP	mean annual temperature in link watershed	Derived from bioclimatic rasters computed using ESOCLIM in the software package ANUCLIM with a 20m DEM as input data
61	CV_TEMP	mean coefficient of variation of mean annual temperature in link watershed	see entry for MEANANN_TEMP
62	MAXWARMP_TEMP	mean maximum temperature of warmest week in link watershed	see entry for MEANANN_TEMP
63	MINCOLDP_TEMP	mean minimum temperature of coldest week in link watershed	see entry for MEANANN_TEMP
64	ANNRANGE_TEMP	mean annual temperature range in link watershed	see entry for MEANANN_TEMP
65	WETQT_TEMP	mean temperature of wettest quarter (any 13 consecutive weeks) in link watershed	see entry for MEANANN_TEMP
66	DRYQT_TEMP	mean temperature of driest quarter (any 13 consecutive weeks) in link watershed	see entry for MEANANN_TEMP
67	WARMQT_TEMP	mean temperature of warmest quarter (any 13 consecutive weeks) in link watershed	see entry for MEANANN_TEMP
68	COLDQT_TEMP	mean temperature of coldest quarter (any 13 consecutive weeks) in link watershed	see entry for MEANANN_TEMP
69	UC_MEAN_TEMP	mean annual temperature in link UCA	see entry for MEANANN_TEMP
70	UCWETQT_TEMP	mean temperature of wettest quarter (any 13 consecutive weeks) in link UCA	see entry for MEANANN_TEMP
71	UCDRYQT_TEMP	mean temperature of driest quarter (any 13 consecutive weeks) in link UCA	see entry for MEANANN_TEMP
72	UCWARMQT_TEMP	mean temperature of warmest quarter (any 13 consecutive weeks) in link UCA	see entry for MEANANN_TEMP
73	UCCOLDQT_TEMP	mean temperature of coldest quarter (any 13 consecutive weeks) in link UCA	see entry for MEANANN_TEMP
74	UC_MEAN_PR	mean annual potential evapotranspiration in link UCA.	
75	SOLDEPTH	mean solum depth in link watershed (m)	Derived from rasters of modeled solum depth extracted from the Soil Hydrological Properties of Australia dataset prepared by Western and McKenzie (2006).
76	UC_SOLDEPTH	mean solum depth in link UCA (m)	as above
77	SOLPAWHC	mean solum plant available water holding capacity in link watershed (mm)	Derived from rasters o modelled soil plant available water holding capacity extracted from the Soil Hydrological Properties of Australia dataset prepared by Western and McKenzie (2006).
78	UC_SOLPAWHC	mean solum plant available water holding capacity in link UCA (mm)	as above

79	K	mean radiometric K value in link watershed (percent)	Derived from rasters of radiometric K obtained from DPI – datasets were compiled from data acquired by the Geological Survey of Victoria (GSV), Geoscience Australia and private companies.
80	UC_K	mean radiometric K value in link UCA (percent)	as above
81	U	mean radiometric U value in link watershed (ppm)	Derived from rasters of radiometric U obtained from DPI – datasets were compiled from data acquired by the Geological Survey of Victoria (GSV), Geoscience Australia and private companies.
82	UC_U	mean radiometric U value in link UCA (ppm)	as above
83	TH	mean radiometric Th value in link watershed (ppm)	Derived from rasters of radiometric Th obtained from DPI – datasets were compiled from data acquired by the Geological Survey of Victoria (GSV), Geoscience Australia and private companies.
84	UC_TH	mean radiometric Th value in link UCA (ppm)	as above
85	STORCOV	mean proportion of link watershed covered by water storages	Derived from rasters of 8 land cover categories extracted from the Modelled Native Vegetation Extent Dataset (NVE2007), kindly provided by Matt White, ARI. The NVE2007 dataset is a model of the current extent of native vegetation across Victoria using time-series LANDSAT imagery together with a number of existing DSE spatial datasets and ground-truthed site data.
86	UC_STORCOV	mean proportion of link UCA covered by water storages	see entry for STORCOV
87	RIP_STORCOV	mean proportion of user-defined riparian zone in link watershed covered by water storages	see entry for STORCOV
88	UC_RIP_STORCOV	mean proportion of user-defined riparian zone in link UCA covered by water storages	see entry for STORCOV
89	NWATBODCOV	mean proportion of link watershed covered by natural waterbodies	see entry for STORCOV
90	UC_NWATBODCOV	mean proportion of link UCA covered by natural waterbodies	see entry for STORCOV
91	RIP_NWATBODCOV	mean proportion of user-defined riparian zone in link watershed covered by natural waterbodies	see entry for STORCOV
92	UC_RIP_NWATBODCOV	mean proportion of user-defined riparian zone in link UCA covered by natural waterbodies	see entry for STORCOV
93	NONATVEG	mean proportion of link watershed without any native vegetation cover	see entry for STORCOV
94	UC_NONATVEG	mean proportion of link UCA without any native vegetation cover	see entry for STORCOV

95	RIP_NONATVEG	mean proportion of user-defined riparian zone in link watershed without any native vegetation cover	see entry for STORCOV
96	UC_RIP_NONATVEG	mean proportion of user-defined riparian zone in link UCA without any native vegetation cover	see entry for STORCOV
97	GRPCCOV	mean proportion of link watershed covered by native grassland/pasture cover	see entry for STORCOV
98	UC_GRPCCOV	mean proportion of link UCA covered by native grassland/pasture cover	see entry for STORCOV
99	RIP_GRPCCOV	mean proportion of user-defined riparian zone in link watershed covered by native grassland/pasture cover	see entry for STORCOV
100	UC_RIP_GRPCCOV	mean proportion of user-defined riparian zone in link UCA covered by native grassland/pasture cover	see entry for STORCOV
101	LOWNATVEG	mean proportion of link watershed covered by low native vegetation cover	see entry for STORCOV
102	UC_LOWNATVEG	mean proportion of link UCA covered by low native vegetation cover	see entry for STORCOV
103	RIP_LOWNATVEG	mean proportion of user-defined riparian zone in link watershed covered by low native vegetation cover	see entry for STORCOV
104	UC_RIP_LOWNATVEG	mean proportion of user-defined riparian zone in link UCA covered by low native vegetation cover	see entry for STORCOV
105	URWBTRECOV	mean proportion of link watershed covered by urban/windbreak tree cover	see entry for STORCOV
106	UCURWBTRECOV	mean proportion of link UCA covered by urban/windbreak tree cover	see entry for STORCOV
107	RIP_URWBTRECOV	mean proportion of user-defined riparian zone in link watershed covered by urban/windbreak tree cover	see entry for STORCOV
108	UC_RIP_URWBTRECOV	mean proportion of user-defined riparian zone in link UCA covered by urban/windbreak tree cover	see entry for STORCOV
109	PLTNTRECOV	mean proportion of link watershed covered by plantation tree cover	see entry for STORCOV
110	UCPLTNTRECOV	mean proportion of link UCA covered by plantation tree cover	see entry for STORCOV
111	RIP_PLTNTRECOV	mean proportion of user-defined riparian zone in link watershed covered by plantation tree cover	see entry for STORCOV
112	UC_RIP_PLTNTRECOV	mean proportion of user-defined riparian zone in link UCA covered by plantation tree cover	see entry for STORCOV
113	DENTRECOV	mean proportion of link watershed covered by dense tree cover	see entry for STORCOV
114	UC_DENTRECOV	mean proportion of link UCA covered by dense tree cover	see entry for STORCOV
115	RIP_DENTRECOV	mean proportion of user-defined riparian zone in link watershed covered by dense tree cover	see entry for STORCOV
116	UC_RIP_DENTRECOV	mean proportion of user-defined riparian zone in link UCA covered by dense tree	see entry for STORCOV

		cover	
117	THINTRECOV	mean proportion of link watershed covered by thin tree cover	see entry for STORCOV
118	UC_THINTRECOV	mean proportion of link UCA covered by thin tree cover	see entry for STORCOV
119	RIP_THINTRECOV	mean proportion of user-defined riparian zone in link watershed covered by thin tree cover	see entry for STORCOV
120	UC_RIP_THINTRECOV	mean proportion of user-defined riparian zone in link UCA covered by thin tree cover	see entry for STORCOV
121	SPARTRECOV	mean proportion of link watershed covered by sparse tree cover	see entry for STORCOV
122	UC_SPARTRECOV	mean proportion of link UCA covered by sparse tree cover	see entry for STORCOV
123	RIP_SPARTRECOV	mean proportion of user-defined riparian zone in link watershed covered by sparse tree cover	see entry for STORCOV
125	UC_RIP_SPARTRECOV	mean proportion of user-defined riparian zone in link UCA covered by sparse tree cover	see entry for STORCOV
126	TRECOV	mean proportion of link watershed covered by any type/combination of tree cover	see entry for STORCOV Obtained by summing URWBTRECOV, PLTNTRECOV, DENTRECOV, THINTRECOV & SPARTRECOV
127	RIP_TRECOV	mean proportion of user-defined riparian zone in link watershed covered by any type/combination of tree cover	see entry for STORCOV Obtained by summing RIP_URWBTRECOV, RIP_PLTNTRECOV, RIP_DENTRECOV, RIP_THINTRECOV & RIP_SPARTRECOV
128	UC_RIP_TRECOV	mean proportion of user-defined riparian zone in link UCA covered by any type/combination of tree cover	see entry for STORCOV Obtained by summing UC_RIP_URWBTRECOV, UC_RIP_PLTNTRECOV, UC_RIP_DENTRECOV, UC_RIP_THINTRECOV & UC_RIP_SPARTRECOV
129	ROAD_DENS	road density in link watershed (km/km^2)	Derived from a raster of road density created from the TR_ROAD_polyline dataset from Vicmap TRANSPORT Note that proposed roads, walking tracks and bicycle paths were excluded as input data to the road density raster.
130	UC_ROADDENS	road density in link UCA (km/km^2)	see above
131	DS_GAUGE	number of gauges along link downstream flow path	Computed using custom ArcInfo script for downstream/upstream tracing
132	DS_DMW	number of dam walls along link downstream flow path	see entry for DS_GAUGE
133	US_DMW	number of dam walls along link upstream flow path	see entry for DS_GAUGE
134	DS_FORD	number of fords along link downstream	see entry for DS_GAUGE

		flow path	
135	US_FORD	number of fords along link upstream flow path	see entry for DS_GAUGE
136	DS_RWF	number of rapids/waterfalls along link downstream flow path	see entry for DS_GAUGE
137	US_RWF	number of rapids/waterfalls along link upstream flow path	see entry for DS_GAUGE
138	DS_TOTLENGTH	total length of streams along link downstream flow path (km)	see entry for DS_GAUGE
139	DS_AVGSLOPE	average slope encountered along link downstream flow path	see entry for DS_GAUGE
140	DS_STDEVSLOPE	standard deviation of slope encountered along link downstream flow path	see entry for DS_GAUGE
141	DS_MAXLSOPE	maximum slope encountered along link downstream flow path	see entry for DS_GAUGE
142	US_AVGSLOPE	average slope encountered along link upstream flow path	see entry for DS_GAUGE
143	US_STDEVSLOPE	standard deviation of slope encountered along link upstream flow path	see entry for DS_GAUGE
144	US_MAXLSOPE	maximum slope encountered along link upstream flow path	see entry for DS_GAUGE
145	DS_AVGFLDW	average floodwidth encountered along downstream flow path	see entry for DS_GAUGE
146	DS_STDEVFLDW	standard deviation of floodwidths encountered along downstream flow path	see entry for DS_GAUGE
147	US_AVGFLDW	average floodwidth encountered along upstream flow path	see entry for DS_GAUGE
148	US_STDEVFLDW	standard deviation of floodwidths encountered along upstream flow path	see entry for DS_GAUGE
149	DS_MAXWARMP_TEMP	mean maximum temperature of the warmest week in watersheds along link downstream flow path	see entry for DS_GAUGE
150	DS_MINCOLDP_TEMP	mean minimum temperature of the coldest week in watersheds along link downstream flow path	see entry for DS_GAUGE
151	DS_DRYQT_TEMP	mean temperature of driest quarter in watersheds along link downstream flow path	see entry for DS_GAUGE
152	DS_WARMQT_TEMP	mean temperature of warmest quarter in watersheds along link downstream flow path	see entry for DS_GAUGE
153	DS_AVGRIPTRECOV	mean riparian tree cover along downstream flow path	see entry for DS_GAUGE
154	DS_STDEVRIPTRECOV	standard deviation of riparian tree cover along downstream flow path	see entry for DS_GAUGE

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APPENDIX

The source(s) and/or derivation of key data input sources for the construction of the ESDV are detailed in the table below.

	Key Data Inputs	Source/Method of Derivation
1	20m DEM of Victoria	Interpolated from 10m-20m contour (EL_CONTOUR) and ground surface point (EL_GRND_SURFACE_POINT) data sourced from Vicmap Elevation using the TOPOGRID algorithm in ArcINFO. Production of the DEM for Victoria involved the main following steps: a) correcting errors in the source data; b) constructing the DEM in 25 overlapping sections/rasters; c) filling sinks to produce a hydrologically correct DEM sections and d) mosaicing the overlapping sections to produce the statewide DEM surface. Evaluation of the DEM output involved: a) creating contours from the new surface and visually comparing them with contours from the input contour data; b) comparing the output drainage coverage with the input stream coverage and finally, c) comparing the new surface with available sections of Vicmap Elevation DEM20.
2	Rasters/grids of bioclimatic variables (various annual and seasonal combinations of rainfall and temperature)	Created using ESOCLIM in the software package ANUCLIM with the 20m DEM as input data.
3	Terrain variables: Topographic Wetness Index (TWI), Multi-resolution Valley Bottom Flatness Index (MrVBF)	TWI raster computed from DEM using ArcINFO AML script. MrVBF raster computed from DEM using purpose-written ArcINFO AML script and additional utilities kindly provided by John Gallant.
4	Edaphic variables: Solum depth, Plant available water holding capacity, Radiometric K, Th and U	Grids of modeled solum depth and plant available water holding capacity were extracted from the Soil Hydrological Properties of Australia dataset prepared by Western and McKenzie (2006). Source data for this data set include the Digital Atlas of Australian Soils from the Bureau of Rural Sciences and estimates of soil properties from McKenzie, N.J., Jacquier, D.W., Ashton, L.J. and Cresswell, H.P., 2000, Estimation of soil properties using the Atlas of Australian Soils, Technical Report 11/00, CSIRO Land and Water, Canberra. Radiometric data measures natural gamma radiation emanating from the earth's surface to a depth of about 30 cm. This gamma radiation is split into four channels - total radioelement count and three naturally occurring elements - potassium (K), thorium (Th) and uranium (U). The varying concentrations and distribution of radiometric potassium, thorium and uranium provide an indication of soil and rock characteristics. The data are used to assist with geological and soils mapping, mineral and petroleum exploration and land managment. Coverage is statewide at varying resolution. Grids of radiometric K, Th and U were obtained from DPI – datasets were compiled from data acquired by the Geological Survey of Victoria (GSV),

		Geoscience Australia and private companies.
5	Land Cover variables	Grids of 8 land cover categories were extracted from the Modelled Native Vegetation Extent Dataset (NVE2007). The NVE2007 dataset is a model of the current extent of native vegetation across Victoria using time-series LANDSAT imagery together with a number of existing DSE spatial datasets and ground-truthed site data. The revised extent layer is a modelled dataset incorporating new and existing models of vegetation cover such as TREE25 (a presence/absence tree cover spatial layer derived from satellite imagery at a scale of 1:25,000) as well as time-series LANDSAT (satellite) imagery, point, line and polygon hydrological features from Vicmap HYDRO, Statewide Forest Resource Inventory (SFRI) data, plantations manually created from aerial photograph interpretation plus site-based training datasets consisting of presence/absence data for each of the structural vegetation types based on many thousands of ground-truthing points and expert validation.
6	Disturbance-related variables: road density	Grid of road density created using ArcGIS Line Density tool with TR_ROAD_polyline dataset from Vicmap TRANSPORT (proposed roads, walking tracks and bicycle paths were excluded) as input data.
7	Instream structures/features: reservoirs, lakes, rapids, waterfalls, stream gauging stations, locks, dam walls and fords	Georeferenced data representing instream structures and features of potential influence on fish were obtained from : a) point, line and polygon data in Vicmap HYDRO (HY_WATER_POINT_point, HY_WATER_STRUCT_LINE_polyline & HY_WATER_AREA_POLYGON_polygon); b) point data in Vicmap TRANSPORT (TR_ROAD_INFRASTRUCTURE_point) and c) Thiess Environmental Services.