# Summary and Review of Approaches to the Bioregionalisation and Classification of Aquatic Ecosystems within Australia and Internationally

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# 1 INTRODUCTION

## 1.1 Background and objectives

The Aquatic Ecosystems Task Group is currently developing a draft national framework for the identification, classification and management of High Conservation Value Aquatic Ecosystems (HCVAE) in Australia. The HCVAE framework is to be consistent with the scientific framework for the National Reserve System and will incorporate principles related to Comprehensiveness, Adequacy and Representativeness (CAR) (TFMPA 1999):

- **Comprehensiveness**: the inclusion of the full range of ecosystems recognised at an appropriate scale within and across each bioregion.
- **Adequacy**: the required level of reservation to ensure the ecological viability and integrity of populations, species and communities.
- **Representativeness**: selection of HCVAE that reasonably reflect the biotic diversity of the aquatic ecosystems from which they derive.

The implementation of these principles when identifying HCVAE across Australia requires the adoption of a nationally consistent bioregionalisation system (to meet the requirement for comprehensiveness) and an aquatic ecosystem classification (to meet the requirement of representativeness). The Aquatic Ecosystems Task Group has commissioned a review of past and current classification and bioregionalisation methodologies to assist in the process of developing and adopting nationally consistent bioregionalisation and aquatic ecosystem classification systems. The aims of this review are to:

- Identify and summarise the key features, purpose and use of past and current aquatic ecosystem bioregionalisations, both nationally and internationally;
- Review, analyse and critique the methods used for developing these bioregionalisations;
- Identify and summarise the key features, purpose and use of past and current aquatic ecosystem classification systems, both nationally and internationally; and
- Review, analyse and critique the methods for developing these classification systems.

This review provides an impartial description and analysis of bioregionalisation and classification systems from aquatic ecosystems. There are no recommendations made for the adoption of any specific classification or bioregionalisation systems or types of systems; this review will inform the Aquatic Ecosystems Task Group as they make such decisions at a later date.

## 1.2 Terminology

A **biogeographic region (e.g. Interim Biogeographic Regionalisation for Australia**, or **IBRA region)** is a region in which the boundaries are determined by vegetation cover, and the earth’s physical features and climate (Natural Resources Management Ministerial Council 2005).

**Bioregion** is defined as a large, geographically distinct area of land and/or water that has assemblages of species and ecosystems forming recognisable patterns within that landscape. A bioregion’s persistence in the landscape reflects the nature of the relationships between physical and biological elements within it, and relatively independent from what is adjacent to it (DEWHA, unpublished).

**Sub-regions** within a bioregion share ecological flows and processes; they also contribute to a unifying set of ecosystem services (DEWHA, unpublished).

**Ecosystems**[1] are unique units comprising a recognisable floristic composition in combination with substrate (lithology/geology layers) and position within the landscape, and including their component biota (where known). An ecosystem map unit should normally be discriminated at a scale of 1:100,000 to 1:250,000. **Regional ecosystems** are ecosystems or a unique unit mapped at some appropriate scale comprising a recognisable floristic composition in combination with substrate (lithology/geology layers) and position in the landscape and including their component biota (Natural Resources Management Ministerial Council 2005). These elements form the bioregional framework as shown in Figure 1.

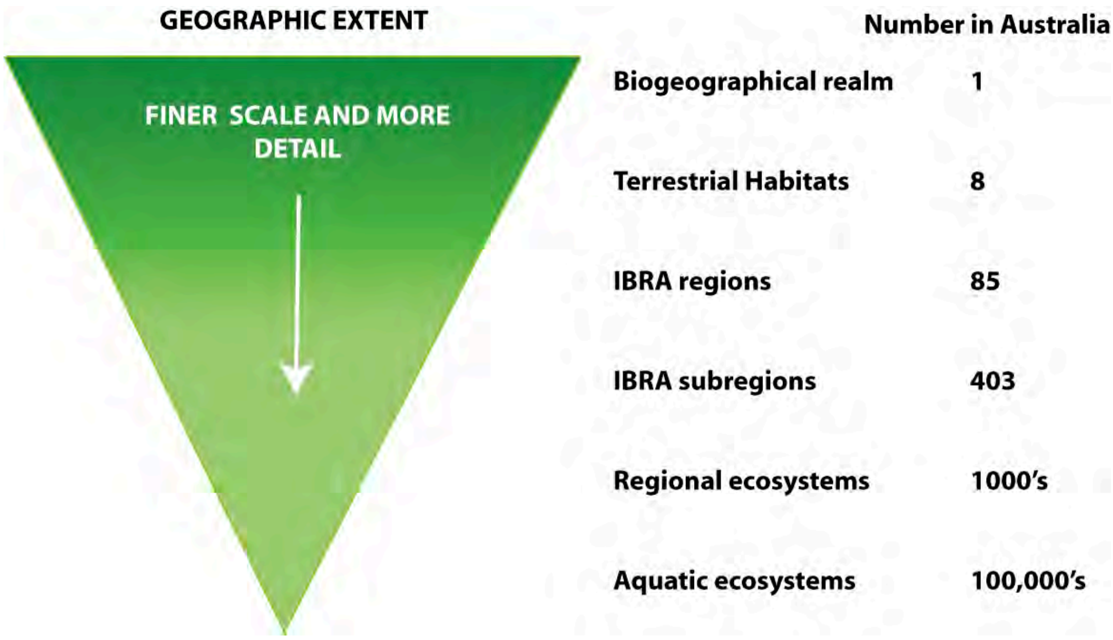


**Figure 1: Conceptual model of the bioregional framework in which IBRA operates. Modified from DEWHA [http://www.environment.gov.au/parks/nrs/science/bioregion-framework/index.html](http://www.environment.gov.au/parks/nrs/science/bioregion-framework/index.html) (as of March 6, 2008).**

**Aquatic Ecosystem -** There is no single definition of wetland or aquatic ecosystems in Australia or internationally. For the purposes of this review, the definition adopted by the Aquatic Ecosystems Task Group has been used:

> *Aquatic Ecosystems are those that depend on flows, or periodic or sustained inundation/ water logging for their ecological integrity (e.g., wetlands, rivers, karst and other groundwater dependent ecosystems, salt marshes and estuaries) but do not generally include marine waters.*

The term **ecoregion** has become widely used in recent times (Omernik and Bailey 1997; Abell et al 2002; Omernik 2004; Mackey et al. 2008) particularly in the international literature. There are many frameworks, however, that use the term without clear definition, leading to misunderstanding and debate over what constitutes an ecoregion. For example the following have been provided as definitions of ecoregion from various sources:

> "*large ecosystems of regional extent that contain a number of smaller ecosystems. They are geographical zones that represent geographical groups or associations of similarly functioning ecosystems*" Bailey (1983, cited by Abell et al. 2002).

[1] Note that the definitions of ecosystem and regional ecosystems as provided by the Natural Resources Management Council (2005) reflect a predominantly terrestrial perspective, despite the claim that the IBRA framework was developed to apply to all terrestrial ecosystems, including freshwater. It is acknowledge by the council that further effort is required to better understand and assess the application of bioregions and sub-regions in relation to freshwater ecosystems and the identification of reservation criteria within this context. It would be expected that refined definitions of ecosystem and regional ecosystem will reflect this in the future.

*"regions of relative homogeneity in ecological systems or in relationships between organisms and their environments"* Omernik (1987, cited by Abell et al. 2002).

*"a relatively large area of land or water containing a geographically distinct cluster of natural communities. These communities:*
- *Share a large majority of their species and ecological dynamics;*
- *Share similar environmental conditions; and*
- *Interact ecologically in ways that are critical for their long term persistence."* WWF - (Abell et al. 2002)

The definitions presented above shows that a range of terms is used in this field and that they are not completely interchangeable. For this reason, the terminology used in each framework reviewed is kept as reported, rather than applying the term bioregion.

## 1.3 Stakeholder feedback

As a component of this review over 60 individuals from government agencies and research organisations were contacted to canvass views on approaches to classification and bioregionalisation and their opinion of the strengths and weaknesses with regard to HCVAE.

The vast number of responses were concerned primarily with regionalisation. While respondents provided details of the classification systems used in their jurisdictions for the classification of aquatic ecosystems, there was little discourse on their use for identifying HCVAE. The overwhelming response, however, was that a national classification system would need to be compatible with the systems currently in use; as most states and agencies did not see change in their currently systems in the near future.

Respondents were asked to comment on any experiences they had had in applying IBRA and how it related to aquatic ecosystems. For most States, the main regionalisation approaches used are IBRA and drainage basins/catchments. However in the majority of cases IBRA has not been used to identify HCVAE. Several States mentioned using IBRA as a high level framework to report on significant aquatic ecosystems, particularly Ramsar and DIWA wetlands, for state of the environment and auditing purposes, and as such feel an additional aquatic ecosystem regionalisation is not required for reporting purposes.

It has been found that States-based regionalisations based on aquatic ecosystems have, in general, not matched well with those of IBRA. The majority of these methods focus on regionalisations based on flowing waters, with few assessments of wetlands (although see Robertson and Fitzsimons 2004) or estuaries. However, as the purposes of the regionalisation are not necessarily directly relevant to conservation planning and identification of high conservation value sites, it is perhaps not surprising that there is little overlap with IBRA regions. As in the international literature, there is considerable difference in opinion between the States as to what constitutes a regionalisation based on the classification of aquatic ecosystem types, with prioritisation method outputs often being considered as regionalisations.

## 1.4 Review structure

This review is presented in two sections: the first considers aquatic ecosystem classification and the second bioregionalisation systems. Classification and bioregionalisation both represent attempts to place artificial boundaries on natural continua (Finlayson and van der Valk 1995; Pressey and Adams 1995). Further, classification and bioregionalisation are in themselves within a continuum based on scale. As a consequence, scale boundaries have been applied to provide definitions of these terms as used in this review. As a guide, the scales of Higgins et al. (2005) have been applied as follows:

- **Classification:** the categorisation of aquatic ecosystems at the scale of individual wetlands, estuaries (< 1 – 10 000 ha) or river reaches (1 – 10 km).
- **Bioregionalisation:** the categorisation of broader areas in the landscape at the scale of groups of wetlands or networks of rivers or larger (> 1000 $km^2$)

# 2 CLASSIFICATION OF AQUATIC ECOSYSTEMS

## 2.1 Context

Classification of aquatic ecosystems is often undertaken for the purposes of inventory, conservation and management planning (Finlayson and van der Valk 1995). There is a large diversity in aquatic ecosystems in terms of form, function, chemical and biological attributes. As with all ecological systems, clear boundaries are difficult to define or delineate and at a landscape scale the diversity in aquatic ecosystems is part of a continuum, rather than a series of discrete units. Some order is required, however, to enable meaningful inventory and facilitate management of aquatic ecosystems: the primary purpose of classification is to group aquatic ecosystems with similar ecological characteristics to facilitate management (Roberts and Butcher 2007). For this review, existing classification systems are considered in terms of their suitability for identifying HCVAE, particularly with regard to ensuring that the HCVAE identified in Australia adequately represent the diversity of wetland types and the biota they support.

It is well recognized that consistent classifications of aquatic ecosystems are required at the national and regional scales both in Australia (Conrick et al in prep; Kingsford et al. 2005) and overseas (Cowardin and Golet 1995). In addition, there have also been calls for a consistent classification system that may be applied at the international scale (Finlayson and van der Valk 1995; Seminiuk and Seminiuk 1997). Despite this, there is no common method for the classification of aquatic ecosystems in Australia or internationally. Rather, there are a large number of different systems used across the globe and within different political jurisdictions within Australia.

It should be remembered that the classification of aquatic ecosystems is distinct from the evaluation of systems (e.g. condition assessment). Classification is considered in terms of typology, whereby the end product is a name and description of the type of aquatic ecosystem. Evaluation in terms of ecosystem condition or environmental, social or economic value is a separate process, not specifically covered by this review.

There have been a number of reviews of aquatic ecosystem classification both in Australia and internationally. More recent Australian reviews include Pressey and Adams (1995) and Roberts and Butcher (2007) for wetlands; Turak (2007); Stein (2007) and Kingsford et al. (2005) for rivers and Calvert et al. (2001) for rivers and estuaries.

This review draws on the results of these previous reviews as well as the scientific literature and considers aquatic ecosystem classification systems in three parts:

1. A description of the common methods of classification with examples from the international literature, together with the general advantages and limitations of each approach;
2. A description of the current systems of aquatic ecosystem classification used in Australia (on a State and Territory basis); and
3. A critique of the various classification systems with respect to ensuring representativeness in the identification of HCVAE at the national, state and regional scales.

## 2.2 Types of classification

The classification of aquatic ecosystems is not a new phenomenon. Shaler developed one of the earliest documented classification systems (1890) for wetlands east of the Rocky Mountains in the United States (Tiner 1999; Figure 2). Since the development of this system there have been numerous advances leading to what Pressey and Adams (1995) described as a “plethora” of classifications.

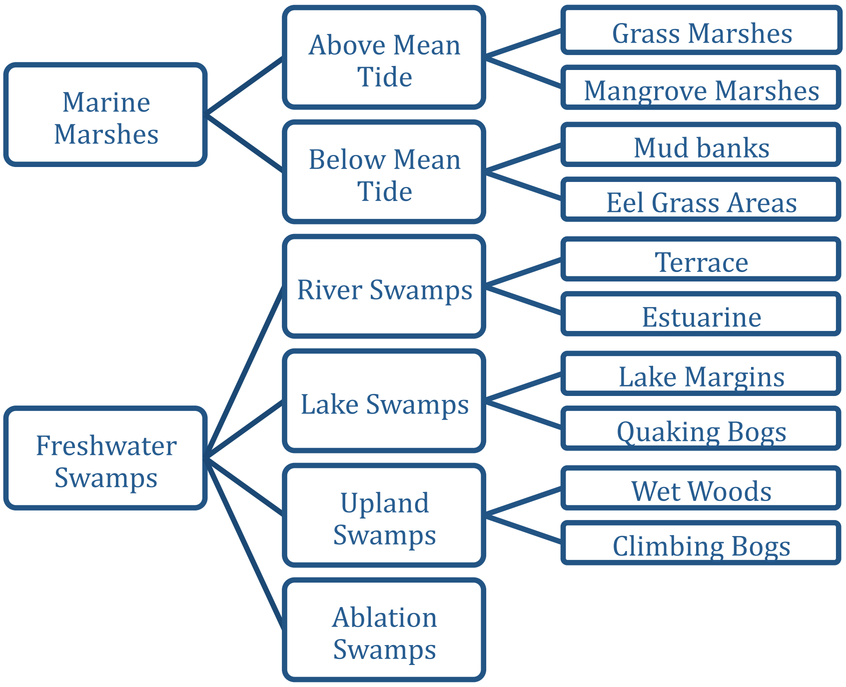


**Figure 2: Shaler's 1890 wetland classification system (adapted from Tiner 1999).**

Systems used to classify aquatic ecosystems fall into two broad categories: those that successively group systems into larger categories on the basis of similarities (bottom-up), and those that divide systems into smaller categories based on differences (top-down). Shaler's system (Figure 2) is an example of a top-down approach, which are more common (Roberts and Butcher 2007) but not necessarily preferable (Robert Pressey, JCU, pers. comm).

There are a large number of variables that may be used to classify aquatic ecosystems, including abiotic factors such as geomorphology and hydrology, and biotic components such as species and communities. In addition, it is common to layer a system of "modifiers", defined by Roberts and Butcher (2007) as "devices for introducing or considering environmental variation without adding complexity to the hierarchical structure". Common modifiers include water quality, adjacent landuse, sediment type and physical alterations such as impounding or excavation.

Although there are seemingly endless combinations of these factors resulting in the "plethora" of systems, Roberts and Butcher (2007) considered that wetland classification systems could be reduced to three main types: criteria-free, mixed criteria and functional. These have been modified and expanded to the following four types for this review:

1. **Unstructured:** unstructured lists based on compilations.
2. **Driver:** based on abiotic drivers of aquatic ecosystems; principally geomorphology and/or hydrology
3. **Biological:** bottom-up classifications based predominantly on species or communities; and
4. **Holistic:** based on a combination of drivers and responses and including both abiotic and biotic components.

### *2.2.1 Unstructured classification systems*

Unstructured classification systems do not have formal hierarchical structures or explicit criteria for the classification of aquatic ecosystems. Drivers, responses and modifiers are not explicitly stated and are not treated separately. Unstructured systems are not geographically restricted, nor are regionalisations considered in the classification of aquatic ecosystems. They are designed so that additional wetland types can be added when required.

Perhaps the best known example of an unstructured classification system is the Ramsar system and those that are derived from it (e.g. the Australian national system for wetland classification).

### Ramsar Wetland Classification System

**Purpose:** Identifying and defining wetlands for the Ramsar Convention.

**Application:** To nominate and list wetlands of international importance. Used by signatory countries for this purpose.

**Key features:** A descriptive list of 43 wetland types under 3 groups; hierarchical but with only two levels; no explicit criteria for designating type; covers rivers, lentic wetlands, estuaries and marine environments.

The Ramsar wetland classification system was originally devised for defining and identifying wetlands of international significance (Seminiuk and Seminiuk 1997), although it is now used to classify wetlands for both conservation and management purposes. The system currently includes 41 wetland types under three broad categories: marine/coastal, inland and human-made (Table 1). Each wetland type is accompanied by a general description. For example, “Freshwater, tree-dominated wetlands; includes freshwater swamp forests, seasonally flooded forests, wooded swamps on inorganic soils”. However, there is little documentation to guide the application of the system and many of the terms used, such as “marsh” “swamp” or “lagoon”, have not been defined. This has led to the following criticism (Seminiuk and Seminiuk 1997; Roberts and Butcher 2007):

- Characteristics of each wetland type are not evident (other than from the name);
- Undefined terms (such as “marsh” and “swamp”) are used in multiple categories leading to potential confusion;
- Attributes used to define wetland types include size, water quality and vegetation, but not systematically;
- Arid-zone wetlands are not adequately represented; and
- Categories are not discrete and often multiple categories can be applied to single wetlands.

Despite this criticism, there have been calls to adopt the system at a global scale (Scott and Jones 1995; Finlayson and van der Valk 1995) to facilitate a common language and better communication between wetland managers.

**Table 1: Ramsar wetland classification (Ramsar 2006)**

| Marine/Coastal Wetlands | Inland Wetlands | Human-made Wetlands |
|---|---|---|
| Permanent shallow marine waters | Permanent inland deltas | Aquaculture (e.g., fish/shrimp) ponds |
| Marine subtidal aquatic beds | Permanent rivers/streams/creeks | Ponds |
| Coral reefs | Seasonal/intermittent/irregular rivers/streams/creeks | Irrigated land |
| Rocky marine shores | Permanent freshwater lakes (over 8 ha) | Seasonally flooded agricultural land |
| Sand, shingle or pebble shores | Seasonal/intermittent freshwater lakes (over 8 ha) | Salt exploitation sites |
| Estuarine waters | Permanent saline/brackish/alkaline lakes | Water storage areas |
| Intertidal mud, sand or salt flats | Seasonal/intermittent saline/brackish/alkaline lakes and flats | Excavations |
| Intertidal marshes | Permanent saline/brackish/alkaline marshes/pools | Wastewater treatment areas |
| Intertidal forested wetlands | Seasonal/intermittent saline/brackish/alkaline marshes/pools | Canals and drainage channels, ditches |
| Coastal brackish/saline lagoons | Permanent freshwater marshes/pools | Karst and other subterranean hydrological systems, human-made |
| Coastal freshwater lagoons | Seasonal/intermittent freshwater marshes/pools | |
| Karst and other subterranean hydrological systems, marine | Non-forested peatlands | |
| | Alpine wetlands | |
| | Tundra wetlands | |
| | Shrub-dominated wetlands | |
| | Freshwater, tree-dominated wetlands | |
| | Forested peatlands | |
| | Freshwater springs; oases | |
| | Geothermal wetlands | |
| | Karst and other subterranean hydrological systems, inland | |

### *2.2.2 Driver based classification systems*

There are many aquatic ecosystem classifications based on fundamental drivers of wetland ecology (geomorphology and hydrology). These drivers are sometimes termed “controlling factors” (Snelder and Biggs 2002) and are based on the assumption that the organisation and function of biotic communities is largely a direct response to the organisation and functioning of the physical environment. Thus, the problem of delineating ecosystem patterns is one of understanding the processes that determine the organisation and functioning of the physical environment. The starting point for many of these systems is a hierarchical model of the assumed causes of ecosystem pattern (Snelder and Biggs 2002).

The proponents of driver-based classification systems argue that by basing classification on fundamental drivers, the categories or classes can be universally applied to geographic regions with different climates and biota; that is, the responses to these drivers may vary in species or communities, but the underlying typology remains the same (Seminiuk and Seminiuk 1995). It is also argued that by basing classification on the drivers of wetland ecology, the resulting categories will represent distinct biological communities, which occur in response to these drivers (Brierley et al. 2002; Snelder and Biggs 2002). There is evidence to support the theory that driver based systems will capture biological patterns. Results from a broad investigation of 889 steams of 29 stream types across the European Union found that the major macroinvertebrate distribution patterns follow climatic and geomorphic conditions and are well distinguished in terms of stream types (Verdonschot and Nijboer 2004). However, there is a growing body of evidence which suggests that classification systems based on landform and hydrology do not necessarily capture biological patterns at the species and genus scales for fish or macroinvertebrates (Paavola et al. 2003; Hawkins; Fielseler and Wolter 2006; Thomson et al. 2004).

Geomorphic and hydrological classifications inevitably require a combination of field assessments complemented by remote sensing. Field assessments need to ensure that adequate representation of each reach or unit is captured and this can be time consuming, limiting the number of sites that can be included (Kingsford et al. 2005). In addition, the low temporal stability of channel form (< 10 years; Naimen et al. 1992) poses problems in consistently applying geomorphically based classifications.

Aquatic ecosystem classification systems typically have a nested hierarchical structure, although this is not always the case (see HGM method and Seminiuk and Seminiuk below) and there are a large number of examples both internationally (Hauer and Smith 1998; Agences de L'Eau 1998; Wimmer et al. 2000; Snelder and Biggs 2002; Rosgen 1994) and from Australia (Seminiuk 1987; Whittington et al. 2001; Brierley et al. 2002). The examples below represent the range of different systems in use, as well as the most commonly utilised.

### Hydrogeomorphic Approach (HGM)

**Purpose:** Functional assessments of wetlands.

**Application:** United States of America, South Africa.

**Key features:** Based on geomorphology and hydrology; hierarchical (not nested), non-prescriptive (does not nominate a type); covers rivers, wetlands and estuaries.

Unlike many other aquatic ecosystem classification systems, the HGM approach was developed for functional assessments rather than wetland inventory. Specifically, HGM is used as a tool for measuring changes in the functioning of wetland ecosystems due to impacts by proposed projects, and restoration, creation, and/or enhancement" (USDA 2008). It has been widely applied throughout the United States of America and formed the basis of the South African Wetland Classification systems (Dini et al. 1998).

The HGM classification system was developed originally by Brinson (1993) and then modified by Smith et al. (1995). Wetland classes and subclasses are based on three attributes:

- **Geomorphic setting:** topographic location within the surrounding landscape;
- **Water source:** precipitation, surface/near surface flow, and ground water discharge; and
- **Hydrodynamics:** direction and strength (hydrologic head) of flow.

The system includes seven classes based on geormorphic setting, three potential dominant water sources and three categories of hydrodynamics. In addition, there are indicators of ecological significance based on water quality and sediment characteristics (Table 2). While hierarchical, the system is not nested, and any of the attributes can be combined to form subclasses. Subclasses and regional subclasses are not limited and users of the system are encouraged to apply the principles of HGM to develop a set of subclasses for their purpose and region. In this manner, the HGM approach is flexible, rather than prescriptive (USDA 2008). It results in a description of wetland types based on function, which has been praised in the literature for its scientific merit (Roberts and Butcher 2007; Smith et al. 2006). However, it does not necessarily lead to placing of wetlands in discrete categories and its applicability to smaller systems (< 1 ha) has been questioned (Kent 2000).

**Table 2: Attributes in the HGM approach for classifying wetlands (adapted from Kent 2000).**

| Geomorphic Setting | Water Source | Hydrodynamics | Ecological Indicators |
|---|---|---|---|
| Riverine | Precipitation | Vertical fluctuations | **Water characteristics:** Suspended sediments Salinity |
| Depressional | Groundwater | Unidirectional flows | |
| Slope | Surface or near-surface | Bidirectional flows | |
| Mineral soil flats | | | **Colour:** Clear Black |
| Organic soil flats | | | |
| Estuarine fringe | | | |
| Lacustrine fringe | | | **pH:** Acid Circumneutral |
| | | | **Nutrient status:** Low Medium High |
| | | | **Soil characteristics:** Mineral Organic |

## River Environment Classification (REC)

**Purpose:** Mapping, inventory and management of rivers.

**Application:** New Zealand.

**Key features:** Based on geomorphology, climate and hydrology; nested hierarchical, covers rivers only.

The River Environment Classification (REC) was developed by Snelder and Biggs (2002) for river management and has been applied to map the rivers of New Zealand. It is based on six "controlling factors" (Snelder et al. 2004):

- Climate – catchment climate;
- Topography – catchment topography;
- Geology – catchment geology;
- Land Cover – catchment landcover;
- Network Position – position of the stream within the catchment network (i.e. stream order); and
- Valley Landform – Landform of the valley the river section is located within.

These are placed within a nested hierarchy (Figure 3) and applied at a range of spatial scales from climatic regions (100,000 km$^2$) to individual river reaches (1km$^2$). The developers claim that patterns in physical and biological characteristics of rivers can be identified using this system and it has been applied to identify fish and invertebrate communities associated with different stream types (Snelder et al. 2004).

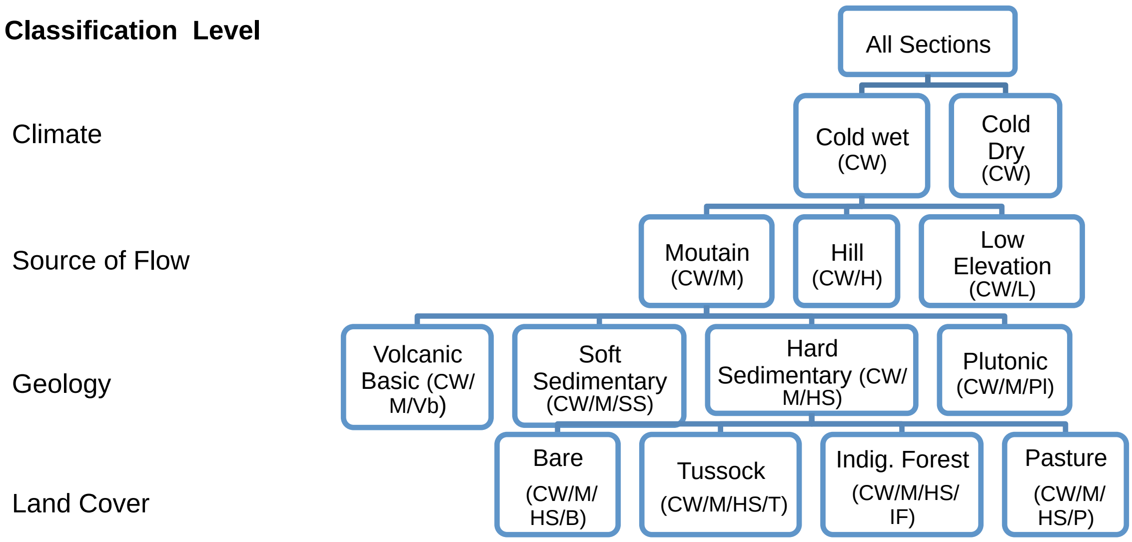


**Figure 3: Example of the River Environment Classification (Snelder et al. 2004) Note: codes in parentheses represent river classifications).**

## Global Classification for Inland Wetlands

**Purpose:** Inventory and mapping

**Application:** Designed to be able to be applied globally, but actual use is restricted to Western Australia.

**Key features:** Based on landform and hydrology; top-down, non-hierarchical, covers rivers, lentic wetlands, but not estuaries or marine systems

Seminiuk and Seminiuk (1995 and 1997) proposed a global system for the classification of inland wetlands based on geomorphology and hydrology. The system is non-hierarchical, using a matrix of landform and hydroperiod to define thirteen wetland categories (Table 3). A series of descriptors based on salinity, wetland size, wetland shape and vegetation pattern are used to further define and describe the wetland types. The system encompasses rivers and inland wetlands but not estuaries, karsts or near shore marine environments. Although proposed as a global system, the references to its application are only from Western Australia (see below). While the purpose of the classification system is not explicitly stated, it was used to map the wetlands over a large part of south-western Australia.

**Table 3: Categories of wetlands based on landform and hydroperiod (Seminiuk and Seminiuk 1995).**

| **Hydroperiod** | **Landform** | | | | |
|---|---|---|---|---|---|
| | **Basin** | **Channel** | **Flat** | **Slope** | **Highland** |
| **Permanent inundation** | Lake | River | - | - | - |
| **Seasonal Inundation** | Sumpland | Creek | Floodplain | - | - |
| **Intermittent inundation** | Playa | Wadi | Barlkarra | - | - |
| **Seasonal Waterlogging** | Dampland | Trough | Palusplain | Paluslope | Palusmount |

### 2.2.3 Biological based classification systems

Although most of the methods used to classify aquatic ecosystems are based on fundamental drivers or a combination of drivers and biota, there are a number of recent classification systems that use only biota (vegetation, fish or macroinvertebrates) as the basis of classification. These “bottom-up” approaches aggregate ecosystems on the basis of similar aquatic communities or species. They use either indicator species or fish or macroinvertebrates assemblage data to determine specific types of aquatic ecosystems. This can be done directly if data are available for a large number of sites (e.g. Eekhout at al. 1997) or by modelling distributions based on a smaller number of reference sites. Biological based classifications have been developed predominantly for rivers and streams (Eekhout at al. 1997; Marchants et al. 2006; Turak and Koop 2008); no examples for lentic systems could be found. Proponents of biological based classifications argue that these systems are the only ones that can accurately reflect the ecology and heterogeneity of communities within aquatic ecosystems (Turak and Koop 2008; Fieseler and Wolter 2006).

The limitations of bottom-up classification systems lie predominantly in the data requirements for their application. They rely on field collection and identification of species from a representative number of reference sites and, given the paucity of biological data for many systems across Australia (and internationally), this makes their application for classification at a national scale difficult (Kingsford et al. 2005). In addition, there is evidence that classification systems based on biological communities (particularly fauna) are not transferable between taxonomic groups. That is, that classifications based on macroinvertebrate assemblages will not necessarily reflect patterns in fish or other biota (Paavola et al. 2003).

There are a number of suggested classification systems for aquatic ecosystems based on biological communities in the scientific literature (Eekhout at al. 1997; Fieseler and Wolter, 2006; Marchants et al. 2006; Turak and Koop 2008). However, most have been developed for bioregionalisations (see below) and there are no instances of any biologically based aquatic ecosystem classification system being adopted and implemented at a national level. Despite this, an example of the biological component of a multi-attribute system developed for NSW rivers is provided below as an indication of how these systems typically are structured.

#### Multi-attribute River Typology (Turak and Koop 2008)

**Purpose:** Condition assessment and conservation planning

**Application:** Developed and applied for the rivers of NSW

**Key features:** A bottom-up, non-hierarchical system that results in a number of different river typologies based on different attributes (abiotic, fish, macroinvertebrate-edge communities, macroinvertebrate-riffle communities).

Turak and Koop (2008) developed a system for river classification in NSW based on a number of attributes, both biotic and abiotic (and could thus be considered as an Holistic classification system – section 2.2.4). As the attributes are not hierarchically linked, it is possible to excise the biologically based ones, presented here as an example. The typology is a bottom-up approach based on the aggregation of site specific data collected in rivers and streams across NSW. Three biological river classifications are presented, based on:

- Fish;
- Macroinvertebrates in stream edges; and
- Macroinvertebrates in riffles.

For each biological group, existing data from the NSW fish database and AUSRIVAS macroinvertebrate family level identifications was analysed to derive river types. This resulted in six river types based on fish, eight river types based on macroinvertebrate edge communities and five river types based on macroinvertebrate riffle communities. The typologies are used separately and each reach is assigned a type under each of the three systems. The results indicated that all western lowland rivers, an area that covers nearly half of the state were essentially of one type, with the majority of the variability in river type exhibited in rivers east of the Great Dividing Range.

The authors state that the strength of this approach is the application of a bottom-up approach using site specific data that can be applied as easily as a top-down approach. However, the classification was based on 322 macroinvertebrate sites and 1500 fish sampling sites, indicating, perhaps that a relatively high level of site specific data is required for its application.

### *2.2.4 Holistic classification systems*

Holistic classification approaches are those that consider abiotic drivers (geomorphology and hydrology) and components (e.g. water quality) as well as biotic components (Sandin et al. 2000). This represents the majority of the systems used for the classification of aquatic ecosystems and as such there are a variety of forms (Cowardin et al. 1979; Farinha et al. 1996; Dini et al. 1998; Ward et al. 2002; Johnson and Gerbeaux 2004; Ball et al. 2006; Thiemes et al. 2007). Some use a combination of physical, chemical and biological attributes with equal weighting and in a mixed format (e.g. the system developed for the United States of America; Cowardin et al. 1979). Others have a functional structure, with drivers at the upper levels of the hierarchy and biological attributes at the finer levels (e.g. the system developed for New Zealand; Johnson and Gerbeaux 2004). It can be argued that those based on functional structure are more scientifically defensible (Roberts and Butcher 2007).

The proponents of these systems argue that they contain all the advantages of the driver based systems in their strong conceptual basis, together with an explicit representation of biological communities (Roberts and Butcher 2007). However, there have been strong criticisms of individual classification systems that use both biotic and abiotic components with regard to their applicability across different geographical regions (Sandin et al. 2000). The examples below are indicative of the range of different systems and the most commonly utilised.

#### Classification of Wetlands and Deepwater Habitats of the United States

**Purpose:** Inventory and mapping of aquatic ecosystems.

**Application:** Implemented across the United States; modified for use in other regions such as Mediterranean Europe, South Africa and Queensland.

**Key features:** A top-down, hierarchical system that uses a mix of geomorphology, hydrology, biota, water quality and sediment characteristics; covers rivers, lentic wetlands, estuaries and near-shore marine environments.

Cowardin et al. (1978) developed a system for wetland classification in the USA that was adopted by the Fish and Wildlife Service for the National Inventory. The primary purpose of the system was for the identification, inventory and mapping of wetlands across the US. This system has formed the basis for a number of others around the globe including the MedWet system adopted by the European Union (Farinha et al. 1996) and the South African Wetland Classification System (Dini et al.1998).

The classification is hierarchical and a top-down approach, dividing wetlands into systems, subsystems, classes, subclasses and dominance types (Figure 4). In addition a series of modifiers based on water regime, salinity, pH, soil and human impacts are used to further describe each of the wetland types. The system includes nearshore marine environments, estuaries, rivers and lentic wetlands and lakes. Although the system has been modified for different continents (Farinha et al. 1996; Dini et al. 1998), it is perhaps the mostly widely accepted and utilised classification for aquatic ecosystems at national and multi-national scales.

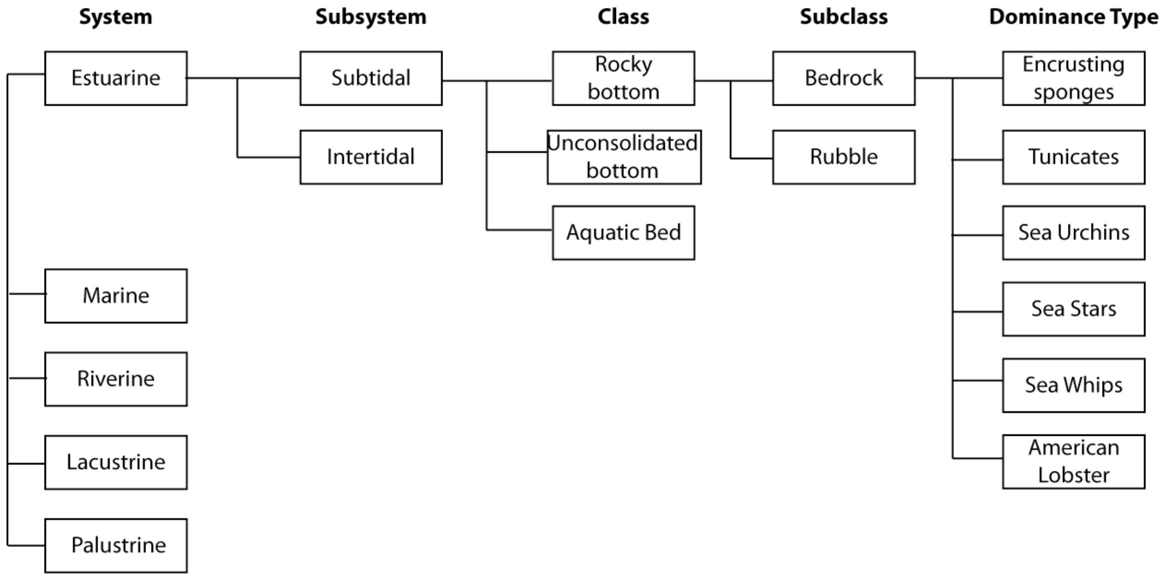


**Figure 4: Example of the hierarchical structure of the Cowardin (1979) classification system. The example shows all five systems, and the complete classification to dominance types for the estuarine, subtidal, rock bottom, bedrock subclass.**

The Cowardin (1979) system was simplified by the USEPA for mapping of wetlands via remote sensing (Ernst et al. 1995). The Environmental Monitoring and Assessment Program (EMAP) aggregated the wetland classes based on dominant vegetation, flooding, water source and distance from river for Palustrine and estuary aquatic ecosystems. This allowed for more broad scale application of the system across the United States.

### The Canadian Wetland Classification System

**Purpose:** Inventory and mapping.

**Application:** Canada.

**Key features:** Based on vegetation, water regime and morphology; non-hierarchical; covers rivers, estuaries and lentic wetlands < 2 m deep.

The Canadian wetland classification system was developed for inventory and mapping of aquatic ecosystems across the country. The system encompasses estuaries, riparian areas and lentic wetlands (< 2m deep) and is structured into wetland classes, forms, subforms and types (Warner and Rubec 1997):

- **Wetland Class**: Wetlands at the class level are recognised on the basis of properties of the wetland that reflect the overall genetic origin of the wetland ecosystem and the nature of the wetland environment. There are five classes: bog, fen, swamp, marsh and shallow water.
- **Wetland Form**: Wetland classes are subdivided into wetland forms based on surface morphology, surface pattern, water type and morphology characteristics of underlying mineral soil. Many of the wetland forms apply to more than one wetland class. Some forms can be further subdivided into subforms.
- **Wetland Type**: Wetland types are subdivisions of the wetland forms and subforms based on physiognomic characteristics of the vegetation communities. Similar wetland types can occur in several wetland classes, while others are unique to specific classes and forms.

The system is non-hierarchical and difficult to illustrate in a condensed form. The classification guide uses a series of keys (similar to taxonomic keys) to aid in the application of the system and to determine wetland types. The system has been developed specifically for the Canadian environment and it is unlikely that it could be meaningfully applied in countries with different wetland forms and vegetation types.

### The New Zealand Wetland Classification System

**Purpose:** Standardise terminology, inventory mapping and management.

**Application:** New Zealand.

**Key features:** Top-down, semi-hierarchical; uses geomorphology and hydrology at the upper levels and vegetation at the lower levels; covers rivers, lentic wetlands, estuaries and near-shore marine environment.

The New Zealand wetland classification system was designed to standardise the terminology associated with wetland types and to facilitate inventory and the management of wetlands in New Zealand (Johnson and Gerbeaux 2004). The system encompasses rivers, estuaries, nearshore marine environments, and lentic wetlands. However, the authors state that it is primarily focussed on inland freshwater systems. The system is semi-hierarchical and is a top-down approach. It comprises of six classification levels (Figure 5), which start at broad geomorphic and hydrological features and end with vegetation associations. The guide provides detailed keys for each level of classification to facilitate identification of wetland type.

The highest level in the hierarchy (hydrosystem) is based on the systems of Cowardin (1979), but expanded to include four addition environments (Inland Saline, Plutonic, Geothermal and Nival). The level of subsystem is descriptive and relates to water regime. The next level of wetland class, is based on substrate, water regime and water quality (nutrients and pH). However it is not hierarchically linked to the upper level, and wetland classes may belong to one or more of the hydrosystems. Wetland form is a descriptive level based on geomorphology. Structural class is based on vegetation structure, or in the absence of vegetation, the dominant ground surface. The final level is based on dominant plant taxa.

The system has been designed with an awareness of spatial scale and mapping and can be applied at different levels. For example, a map scale of 1:100,000 is appropriate for classification to hydrosystem only; 1:50,000 can be used to show wetland class; 1:10,000 is appropriate for vegetation structural classes and 1:5000 is required to portray the entire classification down to the lowest level.

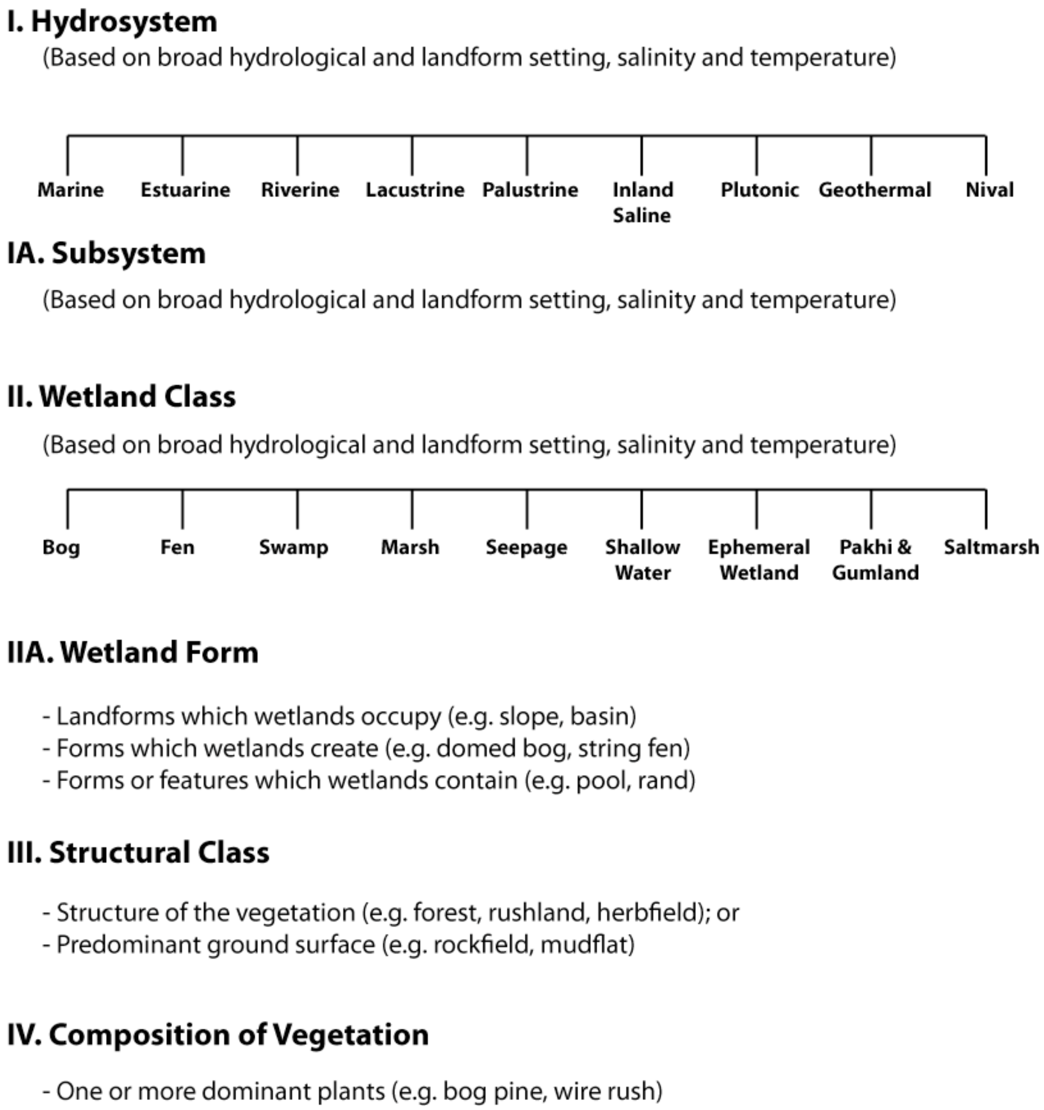


**Figure 5: Overview of the New Zealand wetland classification system (Johnston and Gerbeaux 1997).**

## 2.3 Classification in Australia

There are a number of aquatic ecosystem classifications in use around Australia. The following is a summary of the systems used by state, territory and federal jurisdictions. Although there are a large number of other systems that may have been applied to small areas or regions, or systems that have been developed and not implemented, these are not explicitly summarised here.

### *2.3.1 National and cross-jurisdictional*

#### Directory of Important Wetlands in Australia (DIWA)

**Purpose:** Identifying and defining wetlands for the DIWA.

**Application:** Across Australia for this purpose.

**Key features:** Relatively unstructured list based on Ramsar, modified slightly for Australian conditions; hierarchical but in the absence of explicit criteria for assigning type, covers rivers, lentic wetlands, estuaries and near-shore marine systems.

The wetland classification used by the Federal government is derived from the Ramsar wetland classification system. Like the Ramsar system, it is non-hierarchical and unstructured (Roberts and Butcher 2007). The system is used predominantly for the purposes of identifying important wetlands at the national scale in a similar process to that used for identifying wetlands of international importance under the Ramsar Convention. The Ramsar classification was modified slightly to suit Australian conditions, for example by including non-tidal freshwater forested wetlands and rock pools as wetland types (Larmour 2001).

#### Functional Process Zones

**Purpose:** To provide a framework for reporting on ecological health of rivers.

**Application:** Murray Darling Basin.

**Key features:** Geomorphology based; hierarchical; top-down, covers rivers only.

The Sustainable Rivers Audit (SRA) was developed for use in the Murray Darling basin for the assessment of river health. Under the SRA, a functional, geomorphic approach was developed for the classification of rivers into “type” and to provide a scale for reporting on river health (Whittington et al. 2001). The system operates at two scales: Functional Process Zones (FPZ) and Valley Process Zones (VPZ). FPZs are lengths of a river that have similar discharge and sediment regimes and are defined by gradient, stream power, valley dimensions and boundary material. VPZs operate at large scales and are defined by their sediment transport characteristics. There are eight FPZs within three VPZs (Figure 6) and geomorphic, hydrological and habitat features are described for reach. In addition, conceptual models are used to describe the ecological functioning and characteristic features of each FPZ.

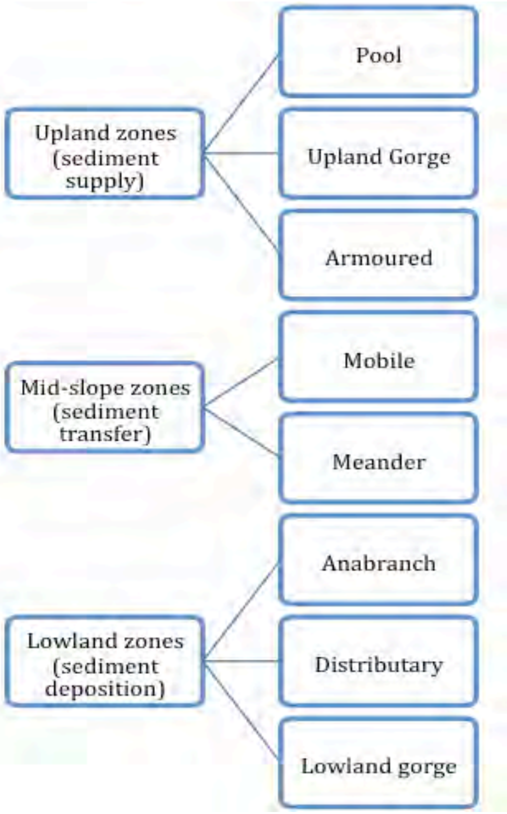


**Figure 6: Structure of the SRA classification showing the three Valley Process Zones on the left and the eight Functional Process Zones on the right (adapted from Whittington et al. 2001).**

### National River Classification

**Purpose:** Classification of rivers for the determination of environmental flows.

**Application:** Proposed for national wide, but not implemented.

**Key features:** Based on geomorphology, climate and water regime; hierarchical, top-down; covers rivers and estuaries.

Calvert et al. (2001) proposed a national classification system for rivers in Australia. The top-down hierarchical system was based on geomorphology, climate and water regime. The system encompasses rivers from headwaters to their discharge into the ocean and included consideration of estuarine and marine reaches. The system recognises 50 river types, although full criteria for classifying each type are not provided. In addition, it appears that this system has not been adopted at by Federal agencies.

### Australian Estuaries

**Purpose:** Biological comparisons of estuaries.

**Application:** Australia wide.

**Key features:** Based on climate and hydrology (tide); hierarchical, top-down; covers estuaries.

Australian estuaries have been classified at the national level according to a physical classification system (Digby et al., 1999), which was designed to:

"*develop a national classification of Australian estuaries based on easily quantifiable biologically important physical characteristics, to enable valid comparisons between biotic communities of different estuaries*".

The system has three hierarchical groupings (Figure 7). The upper level is based on climate, whereby Australia is divided into five groups: tropical, tropical savannah, hot dry, subtropical and temperate. The next level is based on tidal range (high, medium and low) and the final level on intertidal proportion (high medium and low). The resulting 21 classes of estuary were then applied to > 700 estuaries across the country. Validation of the classification indicated that it accounted for over half the variation in saltmarsh and mangrove distributions (Digby et al. 1999).

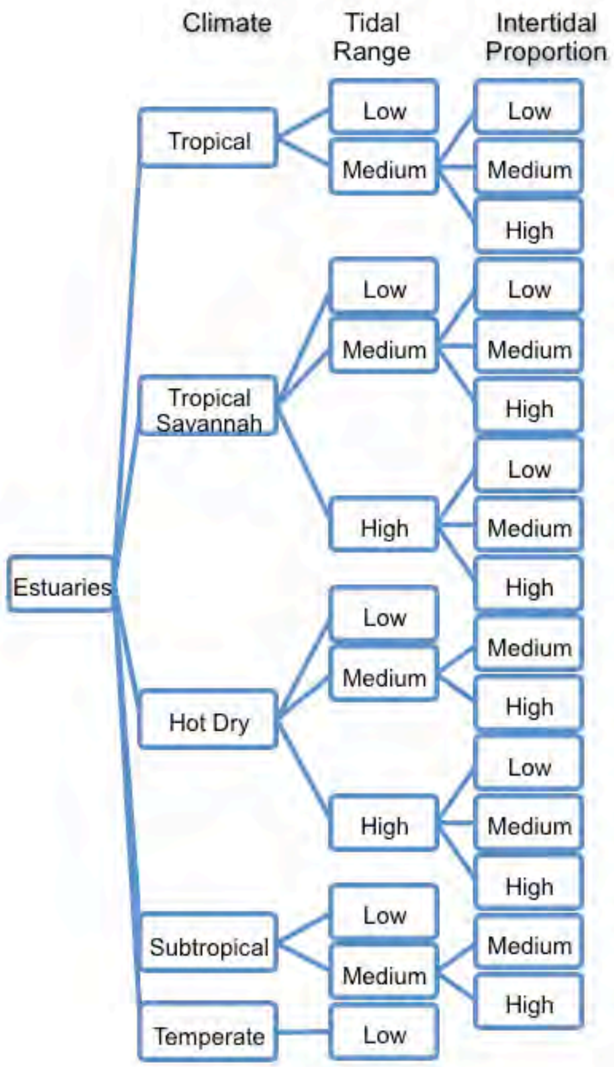


**Figure 7: Hierarchical classification of Australian Estuaries (Digby et al. 2001).**

### *2.3.2 Australian Capital Territory*

The ACT does not have a state based classification system for wetlands or rivers, but follows the Ramsar/DIWA system (Lisa Evans, ACT Parks, pers. comm.).

### 2.3.3 *New South Wales*

**Wetland Classification**

**Purpose:** Wetland management.

**Application:** Designed for NSW, but implemented unevenly across the State.

**Key features:** Hierarchical (two levels), relatively unstructured, descriptive list (although a basic identification key is provided); covers inland lentic wetlands.

The wetland classification system for NSW (Green 1997) provides a framework for the development of wetland management guidelines for specific wetland types and wetland issues. There are two levels to the classification; the upper level is based on geographic region (Coastal, Tableland, Inland), while the lower level is simply termed wetland type and describes 12 types for the State (Table 4).

The form of the system is relatively unstructured, and a key is provided for the identification of wetland type. The system is for non-flowing aquatic ecosystems and while it includes estuaries, rivers are not considered. Although the Department of Natural Resources states that this is the official system for the state (http://naturalresources.nsw.gov.au/water/wetlands_facts_type.shtml, it is rarely used in practice (Roberts and Butcher 2007).

**Table 4: Wetland types for NSW (Green 1997).**

| Coastal | Tablelands | Inland |
|---|---|---|
| Mangrove and saltmarsh swamps | Upland lakes and lagoons | Permanent inland wetlands |
| Estuarine lakes and lagoons | Upland swamps | Inland floodplain lakes and lagoons |
| Dune swamps and lagoons | | Inland floodplain meadows |
| Coastal floodplain swamps and lagoons | | Reed swamps |
| Coastal floodplain forest | | Lignum swamps |
| | | Inland Floodplain forests and woodlands |
| | | Arid wetlands |

**River Styles**

**Purpose:** Identification, mapping and condition assessment.

**Application:** Across NSW, parts of Queensland, South Australia, Tasmania and Victoria; internationally in the United States.

**Key features:** Based on geomorphology, hierarchical (but not nested) bottom-up, non-prescriptive, flexible, covers rivers

River Styles is a geomorphic based system that is used for both classification (identification and mapping) as well as assessment (Brierley et al. 2002). Stage I of the River Styles framework is concerned with classification and this is the only aspect of River Styles considered in this review. The system is hierarchical, but open ended and generic, new styles can be added and the system does not prescriptively apply categories; rather, it seeks to understand the character and behaviour of systems (Brierley et al 2002).

The system classes rivers under three broad groups based on valley setting (Figure 8). Different procedures are then used under each of these groups to identify River Styles. The system is an example of a bottom-up approach and is based on extensive field work to characterise channels (Kingsford et al. 2005).

River Styles has been applied to rivers across the state of NSW as well as in selected catchments in Queensland, South Australia, Tasmania and Victoria. In addition, it has now been applied in parts of the USA (http://www.riverstyles.com/extensions.php).

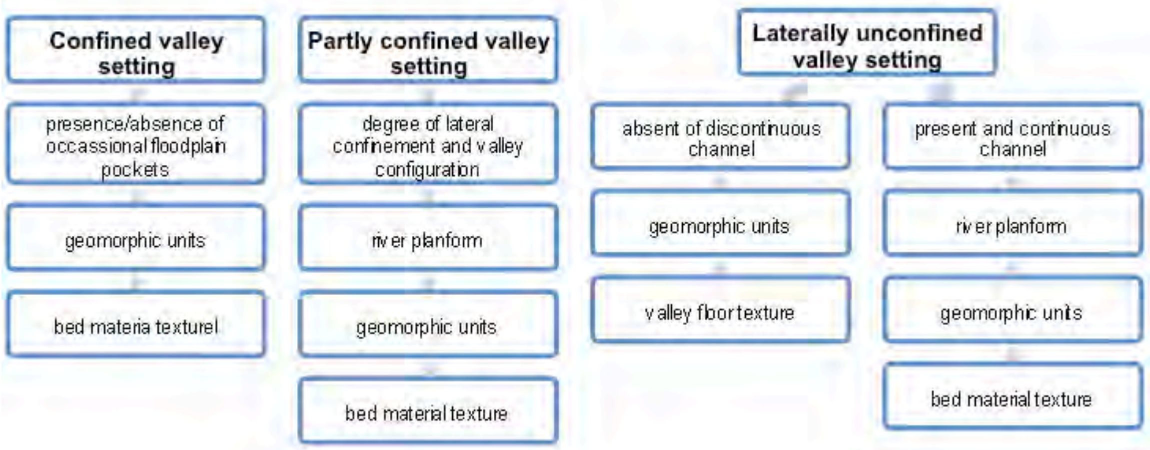


**Figure 8: Procedures used to identify River Styles in different valley settings (Brierley and Fryirs 2005).**

### Multi-attribute River Typlology

**Purpose:** Condition assessment and conservation planning

**Application:** Rivers of NSW

**Key features:** A bottom-up, non-hierarchical system that results in a number of different river typologies based on different attributes (abiotic, fish, macroinvertebrate-edge communities, macroinvertebrate-riffle communities).

The Multi-attribute River Typology (Turak and Koop (2008) has been developed and applied to the rivers of NSW. This is a bottom-up holistic system, which uses a combination of physical and biological attributes. The system encompasses aspects of both classification and bioregionalisation and as such has been considered further in section 3.

## *2.3.4 Northern Territory*

**Purpose:** Inventory and mapping.

**Application:** Arid zone wetlands of the Northern Territory.

**Key features:** Hierarchical, top-down; based on geomorphology and water regime at the upper levels and vegetation at the lower levels; inland rivers and lentic wetlands.

The Northern Territory wetland classification system encompasses rivers and lentic wetlands, but as it was developed for the arid zone wetlands of the Territory, it does not include estuaries and the marine environment. The system is hierarchal, top-down and holistic in that it considers abiotic drivers and biota (vegetation). The system comprises of 71 wetland types that are grouped at the highest level into six categories based on major landform, hydrological function and origin (Duguid et al. 2002):

- Basins (17 types);
- Flats (4 types);
- Channels (21 types);
- Springs (18 types);
- Subterranean (1 types); and
- Artificial (10 types).

At the lower levels, different attributes are used to subdivide each of these categories on the basis of water source, salinity, and permanence. The system was designed to be compatible with the Ramsar classification and divides basins into large (> 8 ha) and small (< 8 ha). In addition, a cross referencing table that matches wetland type from this system with Ramsar wetland types is also provided. The system has been applied to the rivers and wetlands of the arid and semi-arid Northern Territory via a combination of remote sensing and on-ground field surveys.

### *2.3.5 Queensland*

**Purpose:** Inventory and mapping in the first instance, informing assessment and management at later stages.

**Application:** Across Queensland.

**Key features:** Adopts the systems of Cowardin (1979); Hierarchical (top-down and bottom-up); based on climate, geomorphology (upper levels) and vegetation at the lower levels; covers rivers, estuaries, lentic wetlands and near-shore marine environment.

The Queensland wetland classification system encompasses wetlands, rivers and estuaries in a hierarchical, holistic classification (EPA 2005). The system is linked explicitly to its application to the inventory and mapping of wetlands. The initial steps involve determining the geographic location of the wetland with respect to drainage divisions and catchments. Wetlands are then classified into Ecological Systems based on the five systems of Cowardin et al. (1979): marine, estuarine, riverine, lacustrine and palustrine. Modifiers such as water regime, hydrological disturbance and salinity are used to describe the aquatic ecosystems under each system. Wetlands are then intersected with vegetation mapping to result in wetland type. This large number of types is then grouped into the following Management Profiles based on similarities:

- Arid zone lakes (see Table 5 for an example of wetland types);
- Arid-zone swamp wetlands;
- Coastal dune lakes;
- Coastal grass-sedge wetlands;
- Coastal melaleuca swamp wetlands;
- Coastal wet heath/sedgeland wetlands;
- Crater lakes;
- Great Artesian Basin spring wetlands;
- Inland non-arid lakes;
- Inland non-arid swamp wetlands;
- Karst wetlands;
- Mangrove wetlands;
- Palm swamp wetlands;
- Saltmarsh wetlands.

**Table 5: Wetland types under management profile Arid Zone Lakes (EPA 2008).**

| Wetland System | Climatic Zone | Wetland Substrate | Water Type | Water Regime | Landscape Geomorph/ Topography | Vegetation | Wetland Name |
|---|---|---|---|---|---|---|---|
| Lacustrine | Desert | All | Saline | Commonly wet | All | All | Desert commonly wet saline lakes |
| Lacustrine | Desert | All | Fresh | Commonly wet | All | All | Desert commonly wet fresh lakes |
| Lacustrine | Desert | All | All | Periodic inundation | Non-Floodplain | All | Desert periodically inundated non-floodplain lakes |
| Lacustrine | Desert | All | All | Periodic inundation | Floodplain | All | Desert periodically inundated floodplain lakes |

### 2.3.6 South Australia

Although there are regional wetland classification systems used in parts of South Australia (e.g. Pressey 1986) there is no State-wide system of wetland classification, although the Ramsar system is used for wetland inventory (DEH & DLWBC 2003; Roberts and Butcher 2007).

### 2.3.7 Tasmania

**Purpose:** Identification of high conservation value aquatic ecosystems.

**Application:** Tasmania.

**Key features:** Not a classification in the strictest sense; uses multiple variables and modelling to predict aquatic ecosystem character; non-hierarchical, covers rivers, lentic wetlands and estuaries.

Tasmania has developed a wetland classification and prioritisation tool for rivers, wetlands and estuaries - Conservation of Freshwater Ecosystems Values (CFEV) (DPIW 2007). Under this system, aquatic ecosystems are divided into five basic types: rivers, waterbodies, wetlands, estuaries, saltmarshes and karsts (Table 6). Multiple variables are used to characterise individual aquatic ecosystems and river reaches (Danielle Hardie, DPIW, pers. comm.). A predictive model is used to determine pre-European character based on these variables. Assessment based on model extrapolations of on-ground data is then used to describe current condition. Finally, a spatial selection algorithm is used to assess conservation value based on rareness of character and condition.

CFEV agglomerates classification, condition assessment and prioritisation into a single process. This sets it apart from the other classification systems considered in this review. As such it was difficult to compare this with other classification systems where the end point is an ecosystem type. CFEV was, therefore, considered in the critique provided in section 2.4 in terms of the classification systems contained within it.

**Table 6: Variables used to characterize aquatic ecosystems under the CFEV program (DPIW 2007).**

| Rivers | Waterbodies | Wetlands | Estuaries | Karst |
|---|---|---|---|---|
| Fluvial geomorphic river types | Physical class | Physical class | Biophysical class | Physical class |
| Hydrological region | Frog assemblage | Frog assemblage | Biological class | |
| Macroinvertebrate assemblage | Native fish assemblage | Crayfish region | Physical class | |
| Native fish assemblage | Crayfish region | Dominant wetland vegetation type | | |
| Macrophyte assemblage | Tree assemblage | Tree assemblage | | |
| Crayfish region | Tyler class | Tyler class | | |
| Tree assemblage | | | | |

### 2.3.8 Victoria

**Purpose:** Inventory and mapping, assessing changes in wetland type and extent over time.

**Application:** Across Victoria (all wetlands > 1 ha).

**Key features:** Based on water regime, morphology and dominant vegetation; hierarchical (two levels); top-down; inland lentic wetlands only.

Wetlands in Victoria are classified according to a holistic system based on water regime and vegetation (Corrick and Norman 1980). The system comprises of nine wetland categories based on water regime and salinity, which are further divided into 28 subcategories based on dominant vegetation structure (Table 7). It is not a nested hierarchical system, as some subcategories occur in more than one category. In addition, the final "type" is not necessarily applied to an entire wetland, but wetlands can be divided into several different subcategories based on vegetation.

The system has been applied to all wetlands in the State (> 1 ha) for two points in time: 1994 (based on aerial photography and ground survey during the 1970s and 1980s) and 1788 (an estimation of pre-European extent and type).

There is no state-wide classification for rivers in Victoria (Leon Metzeling, Victoria EPA pers. comm.). The Index of Stream Condition, the state-wide assessment method for monitoring river health, divides rivers into reaches based on similar geomorphic and hydrological features. However, this does not assign a river type to the reach, nor are similarities between types explored (Paul Wilson, DSE, pers. comm.).

**Table 7: Wetland categories and subcategories of in Victoria (Corrick and Norman 1980).**

| Category | Subcategory |
|---|---|
| Freshwater Meadows<br>< 0.3m<br>< 4 months of the year inundated | Herb dominated<br>Sedge dominated<br>Red gum dominated<br>Lignum dominated |
| Shallow Freshwater Marshes<br>< 0.5 m<br>< 8 months of the year inundated | Herb dominated<br>Sedge dominated<br>Cane grass dominated<br>Lignum dominated<br>Red gum dominated |
| Deep Freshwater Marshes<br><2 m<br>Semi permanent – dries every 4-5 years | Shrub dominated<br>Reed dominated<br>Sedge dominated<br>Rush dominated<br>Open water<br>Cane grass dominated<br>Lignum dominated<br>Red gum dominated |
| Permanent Open Freshwater<br>< 5 m –shallow, >5 m – deep<br>permanent | Shallow (<5 m)<br>Deep (>5 m)<br>Impoundment |
| Semi permanent Saline wetlands<br><2 m<br><8 months of the year inundated | Salt Pan<br>Salt meadow<br>Salt flats<br>Sea rush dominated<br>Hyper saline lakes |
| Permanent Saline wetlands<br>< 5 m –shallow, >5 m – deep<br>permanent | Shallow (<5 m)<br>Deep (>5 m)<br>Intertidal flats |

### *2.3.9 Western Australia*

Western Australia uses the system developed by Seminiuk and Seminiuk (1995) described in section 2.2.2 to classify rivers and wetlands. This non-hierarchical, driver based system has been applied to wetlands on the Swan Coastal Plain and other isolated areas within the State.

## 2.4 Applicability of Classifications to identifying HCVAE

The purpose of this review is to summarise and critique current classification systems for aquatic ecosystems with regard to their use at the national level for identifying HCVAE. As the criteria for identifying HCVAE in Australia have yet to be finalised, the critique of classification methods is based on the following general criteria:

1. **Hierarchical** – this was a requirement from the project brief that the classification be hierarchical as requested by the Wetlands and Waterbirds Task Group as a criterion.
2. **Scientifically valid** – based on best available science.
3. **Inclusive** – can be applied to wetlands, rivers and estuarine systems
4. **Comprehensive** – is applicable to aquatic ecosystems from all climatic and geographic regions across Australia.
5. **Objective** – can be applied by different people with different values and experiences with the same end result.
6. **Ecologically meaningful** – results in categories with distinct ecological characteristics.
7. **Feasible** - can be applied with the current level of knowledge of aquatic ecosystems, or with minimal additional sampling effort (remote or on-ground).
8. **Compatible** – can be integrated or overlain with current systems of classification used at the national and state levels.

The broad classification methods described previously have been assessed against each of these criteria in turn. In addition, a summary table showing the outputs of this review against all criteria is provided (see section 2.4.9; Table 8).

This desktop review has been based on the available literature and descriptions of classification systems. Objectively testing of the various classification methods considered would require the application of the criteria listed above to wetlands across Australia; such an effort is beyond the scope of this project. The following discussion of criteria should be considered as a general summary to guide the Aquatic Ecosystems Task Group, not as a definitive assessment.

### *2.4.1 Hierarchical*

Each of the broad classification methodologies (driver based, biota based and holistic) can be used to develop a hierarchical classification system for aquatic ecosystems. Specific examples of hierarchical systems from the international literature and the Australian context include:

- Ramsar and DIWA;
- River Environment Continuum (Snelder and Biggs 2002)
- Classification of Wetlands in the USA (Cowardin et al. 1979) and similar systems such as MedWet (Farinha et al. 1996) and the South African Classification System (Dini et al. 1998);
- The New Zealand Classification System (Johnson and Gerbeaux 2004);
- Sustainable Rivers Audit – Functional Process Zones (Whittington et al. 2001);
- Australia Estuaries Classification (Digby et al. 1999);
- Proposed National River Classification (Calvert et al. 2001);
- Northern Territory Wetland Classification (Duguid et al. 2002);
- Queensland Wetland Classification (EPA 2005);
- Classifications within CFEV (DIPW 2007); and
- Victorian Wetland Classification System (Corrick and Norman 1980).

In addition, River Styles (Brierley et al. 2002) could also be considered hierarchical, but not with a formal, nested structure.

### *2.4.2 Scientifically valid*

This criterion is not easy to apply, as all classification systems for aquatic ecosystems have their basis in some aspects of wetland ecology. However, guidance is taken from Roberts and Butcher (2007) who stated that the most scientifically sound classification systems for wetlands were those they termed "functional". This was considered to be those classification systems that considered drivers (hydrology and geomorphology) at the higher levels of the hierarchy. This is supported by Kingsford et al. (2005) who concluded that independent classifications linked via a multi-scaled hierarchical framework such as those that use flow and geomorphology nested under ecoregions were the most valid for conservation purposes.

In terms of this review, the most scientifically robust classification systems were those that utilised concepts of ecosystem function to derive criteria for assigning wetlands into types. This includes some systems that are holistic (i.e. are based on both drivers and biological components) as well as those that are based on drivers of aquatic ecosystem ecology.

Classification systems that best meet this criterion are:

- The Hydrogeomorphic approach (Smith et al. 1995);
- The River Environment Classification (Snelder et al. 2002);
- Global (and Western Australian) classification (Seminiuk and Seminiuk 1995);
- The New Zealand Classification System (Johnson and Gerbeaux 2004);
- Sustainable Rivers Audit – Functional Process Zones (Whittington et al. 2001);
- Australia Estuaries Classification (Digby et al. 1999);
- Proposed National River Classification (Calvert et al. 2001);
- Northern Territory Wetland Classification (Duguid et al. 2002);
- River Styles (Brierley et al. 2002);
- Multi-attribute River Typology (Turak and Koop 2008);
- Queensland Wetland Classification (EPA 2005); and
- Classifications within CFEV (DIPW 2007).

It could be argued that the Cowardin (1979) system and the Victorian wetland classification system (Corrick and Norman 1980) meet this criterion as well. However, Roberts and Butcher (2007) considered neither of these properly includes wetland function. Response and driver variables are mixed under the Cowardin (1979) system, while the Victorian system has little conceptual underpinning and only considers one attribute of hydrology (duration).

### *2.4.3 Inclusive*

All the broad methods for aquatic ecosystem classification (driver based, biota based and holistic) can be inclusive of the full range of aquatic ecosystem types. While it is possible to implement separate classification systems for different types of aquatic ecosystems, it is arguably preferable to adopt or design a classification that encompasses all aquatic ecosystem types. In this way, biases that may occur from the use of separate systems for rivers, lentic wetlands and estuaries can be minimised. As aquatic ecosystems are not always discrete units (e.g. the longitudinal continuum of headwaters to the ocean and the lateral continuum of rivers to floodplains) a single system would avoid duplication and potential mismatches of classifications for systems that exist on the extremes of these continua.

Specific examples of inclusive systems from the international literature and Australian context include:

- Ramsar and DIWA;
- Classification of Wetlands in the USA (Cowardin et al. 1979) and those systems based on this such as MedWet (Farinha et al. 1996) and the South African Classification System (Dini et al. 1998);
- The New Zealand Classification System (Johnson and Gerbeaux 2004);
- Queensland Wetland Classification (EPA 2005); and
- Classifications within CFEV (DIPW 2007).

The Global (and Western Australian) classification (Seminiuk and Seminiuk 1995) and Northern Territory Wetland Classification (Duguid et al. 2002) encompass rivers and lentic wetland, but not estuaries.

### 2.4.4 Comprehensive

Australia is a large and variable continent that contains a range of different climates and landscapes, which in turn affect the type of aquatic ecosystems and the biota that they support. A national aquatic ecosystem classification system needs to be allow for this diversity of aquatic ecosystems in a representative manner. Classifications such as the unstructured lists of Ramsar (and DIWA) have been criticised for their inability to capture the diversity of wetlands, particularly in arid zones (Seminiuk and Seminiuk 1997).

This criterion is difficult to apply and would best be addressed through a series of trials, testing different systems in different locations across Australia. Many of the classification systems developed to date have been region specific (e.g. Cowardin et al. 1979; Johnson and Gerbeaux 2004). While it is possible to purpose-build a classification system using driver or holistic methods, it is unlikely that there are many existing systems that would be immediately fit for this purpose. The exception to this is the national system already in place for Australian estuaries (Digby et al. 1999) and potentially the proposed national river classification (Calvert et al.2001), although the latter has not been tested. In addition, this criterion may be met by the global system of Seminiuk and Seminiuk (1995) that was designed for broad application, and the River Styles system (Brierley et al. 2002) that has been applied across large parts of Australia and now internationally.

### 2.4.5 Objective

A national classification system for aquatic ecosystems will, by necessity, be implemented by a rage of people from different jurisdictions and with different backgrounds. If the resulting inventory is to be used to identify a representative selection of aquatic ecosystems for conservation, then it is essential that the system can be applied consistently. This criterion would also benefit from testing, as it can only be applied in a broad sense by asking the following two questions:

- Does the system result in a discrete type?
- Does the method provide clear, unambiguous criteria to determine type?

The Ramsar (and DIWA) classification systems have been strongly criticised for their lack of guidance on assigning type and the ambiguity of the categories, whereby multiple categories can be applied to single wetlands (Seminiuk and Seminiuk 1997; Roberts and Butcher 2007). Classifications such as the hydrogeomorphic approach (Smith et al. 1995), which do not result in a discrete type, also do not adequately meet this criterion.

Finally, the issue of the application of the classification system at different points in time should be considered. It is recognised that aquatic ecosystems are dynamic and characteristics such as water depth, extent of inundation and biota present can change over short time spans. This has implications for classification and mapping of aquatic ecosystems (Roshier et al. 2001). For example, macroinvertebrate assemblages in wetlands can fluctuate significantly over periods of months (Butcher 2003) and Roux et al. (2002) reported completely different faunal assemblages from two adjacent pools in a single river.

A classification system must be able to be applied consistently at different points in time and achieve the same classification results. This presents problems for systems such as the Victorian wetland classification system (Corrick and Norman 1980), which uses duration and water depth as the only criteria for distinguishing wetland categories. These characteristics would change in most wetlands in periods of drought or flood, resulting in different wetland types being assigned to the same waterbody at different points in time. Similarly, classification systems based solely on fauna can expect to produce different results at different points in time (Kingsford et al. 2005).

Overall, this criterion is likely to be met by all other classification systems included in this review with the exception those mentioned above (Ramsar / DIWA; Hydrogeomorphic and Victorian wetland classification systems).

### 2.4.6 Ecologically meaningful

Classification of aquatic ecosystems is an attempt to put artificial boundaries around continua of natural systems (Kondolf 1995). In order to identify HCVAE that are representative of the biological / biophysical diversity of aquatic ecosystems, the classification system must result in categories that capture this range of diversity. Robertson and Fitzsimons (2004) argue that the ability to establish a protected area system is heavily influenced by the ability to delineate distinct ecological communities.

There are arguments in the literature for and against different classification systems and their ability to capture ecological diversity (Turak and Koop 2008; Jensen et al. 2001 among others). One side of the debate contests that the variability and diversity of aquatic ecosystems is captured in hierarchical classifications that consider controlling and driving attributes (Hurley and Jensen 2001). Ecological theory dictates that the drivers of aquatic ecosystems (climate, geomorphology and hydrology) create the habitat for species (Jensen et al. 2001). Therefore a classification system based on these drivers will also capture the distinct biotic communities that these support. As stated previously, there is evidence to support the theory that driver based systems will capture biological patterns. Results from a broad investigation of 889 steams of 29 stream types across the European Union found that the major macroinvertebrate distribution patterns in European streams follow climatic and geomorphic conditions and are well distinguished in terms of stream types (Verdonschot and Nijboer 2004).

The other side of the debate contests that top-down classification systems based on physical variables do not adequately capture the biological communities of aquatic ecosystems and that classifications based on biological communities are required (Turak and Koop 2008). In support of this there is evidence that driver based classification systems do not capture distinct macroinvertebrate or fish communities (Paavola et al. 2003; Hawkins; Fielseler and Wolter 2006; Thomson et al. 2004). There is also evidence that classification systems based on biological communities (particularly fauna) are not transferable between taxonomic groups. That is, that classifications based on macroinvertebrate assemblages will not necessarily reflect patterns in fish or other biota (Paavola et al. 2003).

The ability of driver based classification systems to capture distinct biological communities is perhaps a question of scale. There is no doubt that at very broad scales, there are differences between biological communities. For example, tropical species assemblages are different from those in temperate regions. Similarly, the biological communities of headwater streams are distinct from those in estuaries. However, when biological communities are assessed at a finer scale, often species assemblages are different over small spatial and temporal scales (Butcher 2003; Roux et al. 2002).

Between these two sides of the debate, lies a third view that integrates attributes of both. There is support for holistic classification systems that integrate disciplines and synthesise concepts (Hurley and Jensen 2001). The most comprehensive classification systems are those that link aquatic patterns and processes with the driving variables of different geomorphic setting (Frissell et al. 1986). Therefore, this criterion is arguably best met by classification systems that consider physical drivers linked with biological attributes.

Classification systems considered in this review that best meet this criterion are:

- The New Zealand Classification System (Johnson and Gerbeaux 2004);
- Classification of Wetlands in the USA (Cowardin et al. 1979) and those systems based on this such as MedWet (Farinha et al. 1996) and the South African Classification System (Dini et al. 1998);
- Multi-attribute River Typology (Turak and Koop 2008);
- Northern Territory Wetland Classification (Duguid et al. 2002);
- Queensland Wetland Classification (EPA 2005);
- Classifications within CFEV (DIPW 2007); and
- Victorian wetland classification system (Corrick and Norman 1980).

### 2.4.7 Feasible

This criterion considers the relative difficulty in applying the system across Australia. The degree to which a classification system is applied (in terms of inventory and mapping) can have a significant effect on the ability to adequately identify ecosystems of high conservation value (Robertson and Fitzsimmons 2004). Bottom-up approaches to aquatic ecosystem classification depend on data at fine scales and usually require detailed field investigations. For example, Kingsford et al. (2005) suggested that a River Styles assessment could take a full day per river reach. This severely limits the application of such approaches in terms of spatial distribution.

Remote sensing and modeling can be used to extend the coverage of field collected data. The CFEV program in Tasmania employs this technique by use of fuzzy logic and spatial selection algorithms (Danielle Hardie, DIPW, pers. comm.). Similarly, the application of the geomorphic classification developed by Seminiuk and Seminiuk (1995) in Western Australia has used a process of remote sensing and ground-truthing (Michael Coote, DEC, pers. comm.). Although this is a well accepted practice, it is not without its problems. The development and application of predictive models depends on representative field data and high quality imagery (Kingsford et al. 2005). Data that does not properly capture the range of variability and the characteristics of aquatic ecosystems can result in flawed or misrepresentative classifications when extrapolated over large areas. Another consideration is the logistics of access to remote areas of Australia. Brooks et al. (2005) were limited to the use of Landsat imagery for the classification system they could apply to streams in Northern Queensland based on the remote nature of the location. This is likely to hold true for other areas in the country where access is not feasible because of physical or financial constraints.

Top-down approaches allow for application of a classification to different levels in the hierarchy based on available data. The New Zealand wetland classification system (Johnson and Gerbeaux 2004) makes this explicit, by indicating the appropriate mapping scales for each level in the classification system. The Nature Conservancy (Higgins et al. 2005) adopts a similarly flexible approach that can be applied to different levels based on available data. For example, the application of this classification system in North America, where fine scale data were available, led to classification to the macro-habitat scale. In areas of South America, where there was a lack of on ground data, aquatic ecosystems were classified to a higher level in the hierarchy.

For implementation over large areas of Australia, this criterion was considered to be best met by classification systems that were hierarchical and top-down:

- Ramsar and DIWA;
- River Environment Continuum (Snelder and Biggs 2002);
- Classification of Wetlands in the USA (Cowardin et al. 1979) and those systems based on this such as MedWet (Farinha et al. 1996) and the South African Classification System (Dini et al. 1998);
- The New Zealand Classification System (Johnson and Gerbeaux 2004);
- Australia Estuaries Classification (Digby et al. 1999);
- Proposed National River Classification (Calvert et al. 2001);
- Northern Territory Wetland Classification (Duguid et al. 2002);
- Queensland Wetland Classification (EPA 2005);
- Classifications within CFEV (DIPW 2007); and
- Victorian Wetland Classification System (Corrick and Norman 1980).

### 2.4.8 Compatible

As described in section 2.3 above, numerous classification systems have been applied aquatic ecosystems in Australia. Many of these have been developed and implemented over relatively long periods and represent a substantial investment in terms of inventory, mapping and aquatic ecosystem management. Although there is a need for a national system, it must be recognized that this will need to make good use of existing systems and data. In addition, the system adopted (or developed) will need to be compatible with the Ramsar system for nominating wetlands of international importance. This was recognised by the newly developed Northern Territory Wetland Classification (Duguid et al. 2002), which provides a look-up table for cross referencing their wetland types with Ramsar categories.

A complete analysis of the compatibility of each of the systems used in Australia with each other and the Ramsar (DIWA) system is beyond the scope of this review and a matter that is being considered under the Australian Wetland Inventory.

### *2.4.9 Summary*

A summary of each of the classification methods and systems considered in this review, against the eight criteria is provided in Table 8.

**Table 8: Summary of criteria against classification methods and systems (X indicates the classification system meets the criterion). Note that this has been a subjective assessment and that criterion 8 could not be applied.**

| **Classification systems** | **Criteria for review** | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** |
| Unstructured | X | | X | X | | | | |
| Driver based | X | X | X | X | X | | | |
| Biologically based | X | | X | X | X | | | |
| Holistic | X | X | X | X | X | | | |
| Ramsar / DIWA | X | | X | | | | X | |
| Hydrogeomorphic | | X | | | | | | |
| River Environment Classification | X | X | | | X | | X | |
| Global & WA (Senimiuk &Seminiuk 1997) | X | X | | X | X | | | |
| USA (Cowardin et al. 1979) | X | | X | | X | X | X | |
| Canada (Warner & Rubec 1997) | | | | | X | | | |
| New Zealand (Johnson & Gerbeaux 2004) | X | X | X | | X | X | X | |
| SRA – FPZ (Whittington et al. 2001) | X | X | | | X | | | |
| National River Classification (Calvert et al. 2001) | X | X | | X | X | | X | |
| Australia Estuaries (Digby et al. 1999) | X | X | | X | X | | X | |
| NSW Wetlands (Green 1997) | X | | | | | | | |
| River Styles (Brierley et al. 2002) | | X | | | X | | | |
| NT Arid Wetlands (Duguid et al. 2002) | X | X | | | X | X | X | |
| Multi-attribute River Typology (Turak Koop 2008) | | X | | | X | X | | |
| Queensland Wetlands (EPA 2005) | X | X | X | | X | X | X | |
| Classifications within CFEV (DIPW 2007) | X | X | X | | X | X | X | |
| Victoria Wetlands (Corrick and Norman 1980) | X | | | | | X | X | |

Key to Criteria:

1. **Hierarchical** – this was a requirement from the project brief that the classification be hierarchical as requested by the Wetlands and Waterbirds Task Group as a criterion.
2. **Scientifically valid** – based on best available science.
3. **Inclusive** – can be applied to wetlands, rivers and estuarine systems
4. **Comprehensive** – is applicable to aquatic ecosystems from all climatic and geographic regions across Australia.
5. **Objective** – can be applied by different people with different values and experiences with the same end result.
6. **Ecologically meaningful** – results in categories with distinct ecological characteristics.
7. **Feasible** - can be applied with the current level of knowledge of aquatic ecosystems, or with minimal additional sampling effort (remote or on-ground).
8. **Compatible** – can be integrated or overlain with current systems of classification used at the national and state levels.

# 3 BIOREGIONALISATION OF AQUATIC ECOSYSTEMS

## 3.1 Context

Regionalisation is a form of classification, and is sometimes referred to as spatial classification, or the classification of areas (Loveland and Merchant 2004). The purpose of regionalisation is to group entities of interest on the basis of properties they have in common, and to reduce the complexity of the entities of interest to a manageable and understandable framework.

Biogeographical regionalisations have become a common tool in environmental management in recent years (Omernik and Bailey 1997; Olson et al. 2001; Loveland and Merchant 2004; Mackey et al. 2008) shifting the focus from an emphasis on individual natural resource management in which assessment, research and monitoring were based on subjects such as forests, rivers, wetlands, wildlife, fish, and agriculture (Omernik and Bailey 1997; Omernik 2004). With this shift there has come a plethora of methods that utilise variations of the theme of biogeographic regionalisation for varying purpose. Unfortunately, this has resulted in a lack of agreement on the theoretical basis of regionalisations, and dispute over definitions, accepted methodology and the delineation of boundaries (Omernik 2004; Mackey et al. 2008).

Loveland and Merchant (2004) note that biogeographic regionalisation is based in two sciences: ecology and geography. As with the classification of aquatic ecosystems, there can be “bottom-up” and “top down” approaches to regionalisation and there is currently no single approach that can be applied to all situations. It is not a straightforward exercise to compare different regionalisations (often developed using widely different methods) to test if they produce similar groupings. If similarities in groupings are found this would suggest that there is validity in the underlying pattern of environmental gradients detected, that they reflect real divisions, and provide confidence for their use at the policy level (Bunce et al. 2002). However comparing different methods will always provide some degree of dissimilarity in the groupings – they are after all different methods (Bunce et al. 2002).

There is considerable difference of opinion regarding the definition of terms used in relation to biogeographic regionalisations, and although there are increasing number of examples in which there has been consensus on delineation of ecoregions, there is still considerable confusion and misuse of terms. The following are the main reasons why this is the case (modified from Omernik 2004):

1. Disagreement on the definition of ecosystems;
2. The complexity of the nature of ecoregions and ecoregion boundaries;
3. Bias toward particular characteristics;
4. Inability or reluctance to embrace a holistic ecosystem concept and preoccupation with specific objectives and reductive methods;
5. Disagreement on whether to use quantitative (rule based) or qualitative (weight of evidence) approaches;
6. Disagreement over whether catchments or watersheds comprise ecoregions; and
7. Investment in existing frameworks and reluctance to change.

Ecosystems are complex and dynamic, both spatially and temporally, leading some to claim that ecosystems exist only as a conceptual unit, the delineation of which will vary with varying goals or purposes (Omernik 2004). How ecosystems are defined and how best to define units of regionalisation are contentious issues, as is the delineation of boundaries and debate on whether catchments equate to ecoregions (Omernik 2004).

**The need for an aquatic bioregional framework**
Whilst the need to conserve aquatic biodiversity is well recognised (e.g. Abell et al. 2002; Saunders et al. 2002; Allan and Flecker 2003; Tait et al. 2003; Robertson and Fitzsimons 2004; Nevill and Phillips 2004; Higgins et al. 2005; Kingsford and Nevill 2005; Kingsford et al.

2005; Nevill 2005; Silk and Ciruna 2005; Dudgeon et al. 2006; Finlayson 2006; Nevill 2006; Phillips and Butcher 2006; Abell et al. 2007; Linke et al. 2007; Thieme et al. 2007; Moilanen et al. 2008), there is ongoing debate both within Australia and internationally as to how best to achieve this. While some argue that there is no need to develop a framework for regionalisation of aquatic ecosystems (personal comments received via stakeholder consultation), others feel it is a critical first step for successful conservation planning in which aquatic ecosystems are adequately covered (e.g. Tait et al. 2003; Kingsford et al. 2005).

Calls for spatial frameworks that specifically address aquatic ecosystems have come from the recognition that aquatic biodiversity has unique traits when compared with terrestrial biodiversity. With regard to systematic conservation planning (Margules and Pressey 2000) the main difference for terrestrial and aquatic ecosystems is the spatial configuration of protected areas, riverine ecosystems in particular are highly connected both longitudinally and laterally (Abell et al. 2002; Saunders et al. 2002; Linke 2006; Phillips and Butcher 2006; Linke et al. 2007; Roux et al. 2008). Connectivity is a central feature which distinguishes aquatic and terrestrial ecoregion/bioregion conservation, as aquatic biodiversity patterns do not necessarily relate well to terrestrial vegetation patterns often used in bioregionalisations (Abell et al. 2002). Modelling by Moilanen et al. (2008) showed that connectivity had a major influence on areas proposed for freshwater conservation in the North Island of New Zealand, with increased fragmentation occurring if it was not taken into consideration.

In Australia, the National Reserve System Task Group (NRSTG) is working to better include the requirements for protecting comprehensive, adequate and representative (CAR) samples of aquatic ecosystems in the National Reserve System (NRS). The *Directions for the National Reserve System: A Partnership Approach* (Natural Resource Management Ministerial Council 2005) details 11 short-term biodiversity targets for the NRS CAR system, with Direction 7 specifically addressing freshwater biodiversity:

> *Direction 7: The current understanding of freshwater biodiversity in relation to CAR to be reviewed and agreed approach finalised, which may include future amendments to the NRS Scientific Guidelines, to ensure freshwater ecosystems are appropriately incorporated with the NRS.*

This was to include:

> *Provision of refined ecosystem mapping at the national level, building on regional mapping. Of particular significance is mapping to assist with planning priorities so that ecosystems reduced to <30% and <10% of their original areas can be identified*.

Kingsford et al. (2005) called for a nationally consistent collection of information for the assessment and identification of HCVAE, claiming that an agreed spatial framework is essential for undertaking national assessments. The Wetlands and Waterbirds Task Force (WWTF) recently agreed that for the purpose of a strategic framework for identifying new Ramsar sites and applying Ramsar listing criteria, an appropriate bioregionalisation should (Ken Morgan, DEWHA, pers. comm.):

- Be relevant to aquatic ecosystems;
- Consist of a suitable number of bioregions that allows identification of the most outstanding examples;
- Be based on available data so that the system can be readily applied across Australia; and
- Be relatively simple to apply so that it is easy to determine which bioregion a wetland sits in.

The choice of the spatial framework for aquatic biodiversity is often seen as a choice between using either a terrestrial based ecoregion/bioregion or a water basin approach, with valid arguments for both. However, it is not necessarily a mutually exclusive choice (Omernik and Bailey 1997). Bryce and Clarke (1996) claim that ecoregions provide a natural complement to drainage basins, with both being required to adequately explore spatial patterns and management options. Large river basins that include areas of relief with clear upland and lowland regions are considered too heterogeneous to clearly explain aquatic ecosystem patterns. Small headwater streams will be more similar to other headwater streams in different basins within the one ecoregion than to lowland streams in the same basin but within a different ecoregion (Bryce and Clarke, 1996).

This review draws on unpublished material and the scientific literature to consider biogeographical regionalisations that either directly consider aquatic ecosystems, or have the potential to be relevant when considering aquatic ecosystems and is presented in three parts:

1. An introduction to the different types of regionalisations, and methods of development;
2. A description of the past and current regionalisations by type including examples from the international literature and Australia; and
3. A critique of the various regionalisation systems with respect to identification of HCVAE at the national, state and regional scales.

Actual comparison of the outputs of different regionalisations is beyond the scope of the present review. The definitions presented in section 1 show that a range of terms are used in this field and that they are not completely interchangeable. For this reason, the terminology used in each framework reviewed is kept as reported, rather than applying the term bioregion.

## 3.2 Types of regionalisations

Fundamental questions that must be answered when mapping ecological regions are: what ecological phenomena are important in defining regions and, how should the boundaries of the regions be determined (McMahon et al. 2004)? Mackey et al. (2008) identify three basic themes for bioregionalisations relating to species distributions, environmental variables such as climate, geology etc, and ecosystem responses such as productivity (Proches 2008) which we have modified and expanded to the following four types for this review:

1. **Biological:** based on bottom up approaches which use primary attributes relating to species, communities and phylogenies. These typically include measures of species richness, endemism, community composition and structure;
2. **Ecosystem drivers:** based on environmental variables such as climate, water availability, soil and air characteristics. Disturbance such as flooding and fire are also considered drivers;
3. **Ecosystem responses:** based on variables such as variation in vegetation productivity; and
4. **Holistic:** based on a combination of driver and response variables and including both abiotic and biotic components.

There are significant differences of opinion on how to undertake biogeographical regionalisations, with approaches being either qualitative or quantitative (Loveland and Merchant 2004; McMahon et al. 2004; Mackey et al. 2008). There are pros and cons to both approaches, and it is crucial that whichever is used be accompanied by a statement of purpose and the temporal and spatial scales of interest (McMahon et al. 2004).

Qualitative methods (e.g. IBRA) predominantly rely on weight of evidence and expert opinion, and are also referred to as integrated survey approaches (Mackey et al. 2008). McMahon et al. (2004) suggest qualitative approaches are conceptually similar to quantitative approaches as they are a type of multivariate analysis, with multiple maps considered synoptically to uncover patterns of interest. The quality of the input data, choice of variables and experience and knowledge of the map makers influence the final product. The outcome relies heavily on expert judgment, and whilst sometimes considered subjective they are not necessarily arbitrary (McMahon et al. 2004). A criticism of qualitative methods is that they are not replicable, although this is not necessarily true as long as decision rules and other expertise used in the methods are clearly documented (McMahon et al. 2004). Methods which include numerical analysis but rely on expert opinion to establish region boundaries are considered qualitative (e.g. Wells et al. 2002).

Quantitative methods (e.g. Mackey et al. 2008 environmental domain regionalisation) also rely on multiple lines of evidence and the emergence of patterns from the analysis component data. They have evolved with the advent of GIS and multivariate techniques and are also referred to as numerical methods. Advocates of this approach proclaim them better able to produce regions that are reproducible and objective (McMahon et al. 2004; Mackey et al. 2008). Expert judgment is still applied in this type of approach when choosing input data, clustering techniques and interpretation of boundaries. Arguments for this type of approach revolve around the ability to deal with large, complex data sets and that eigenanalysis and clustering techniques can reveal patterns where qualitative approaches can not (McMahon et al. 2004).

The key elements used in reviewing bioregionalisations from Australia and internationally are shown in Figure 9. The allocation of methods to a bioregionalisation type has been based on the primary variables used to describe the regions. For example, if some driver attributes are used as base layers but the final description of the regions is based on floristic composition, then this is considered a biological approach. Not every regionalisation encountered has been reviewed; examples have been provided to account for the range of methods in use, both in Australia and internationally.

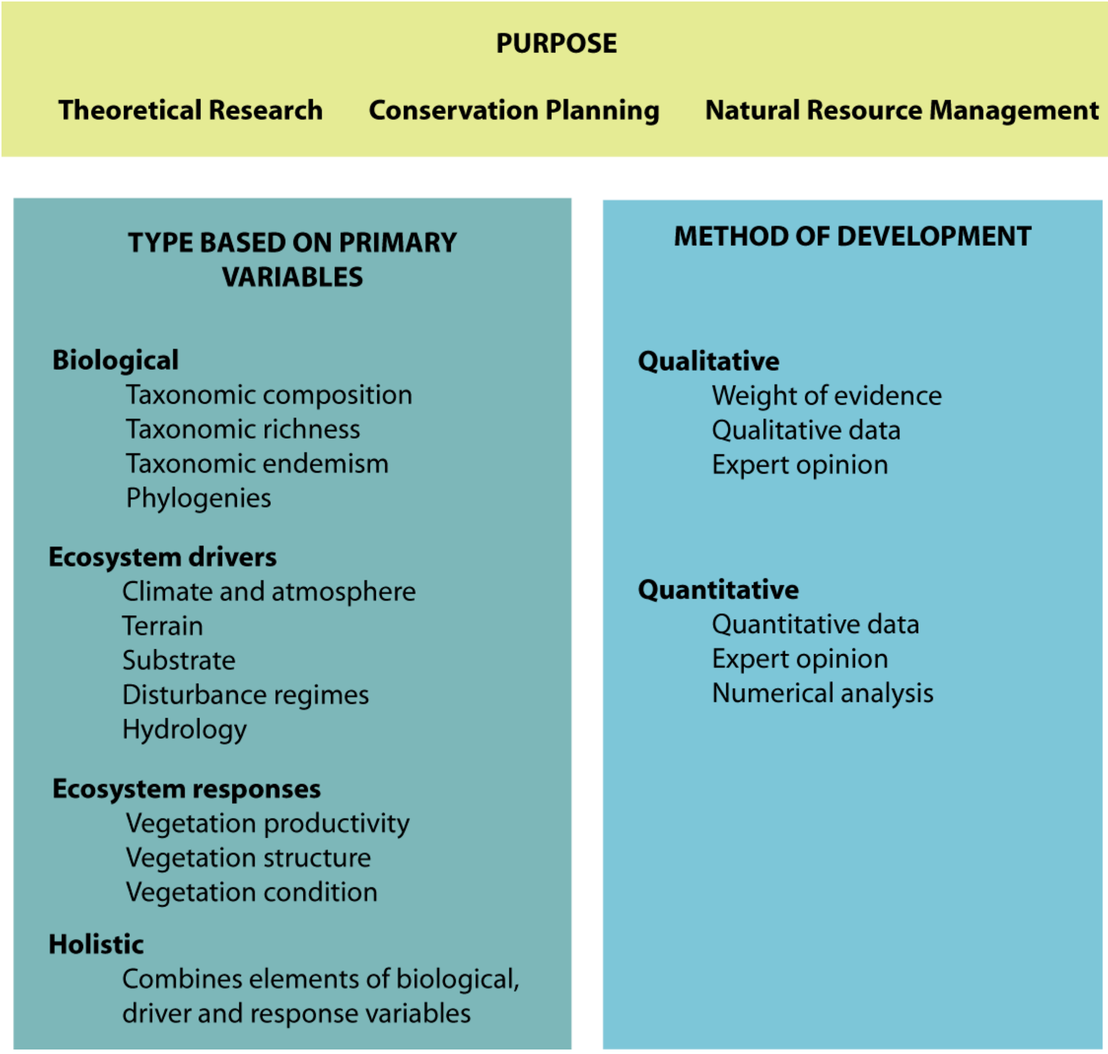


**Figure 9: Purpose, type and approach to biogeographic regionalisations (modified from Mackey et al. 2008). The purpose for which regionalisations can be developed is not an exhaustive list and only represents the three most relevant to the current review).**

## 3.3 Biological regionalisations

Regionalisations in this group are built on biological variables, which include primary data on species distributions and secondary variables such as richness and endemism. Variables can be inferred from molecular data related to phylogenies and genetic structures (Mackey et al. 2008) and most of the regionalisations of this type include consideration of ecosystem driver variables such as climate as well. Approaches included in this section are those that use biological data as the primary variables, for example those which use species distribution data. Whilst many biological regionalisations face data limitations, this does not mean the bottom up approach is not a valid approach.

One criticism of this type of approach to regionalisation for biodiversity conservation relates to the applicability of regions based on single taxonomic groups for other aquatic biota. For example Growns (in review) reviewed the regionalisations produced using macroinvertebrate, fish, frog and macrophyte data for NSW and found that each classification differed and that a classification based on the combined data failed to match any of the individual regionalisations. Growns (in review) also showed that the importance of the underlying environmental gradients that described species turnover varied among the four taxonomic groups. Burgess et al. (2006) analysed species values (richness and endemism) and non-species values (migrations, intact assemblages) and threats, producing a single assessment of conservation priorities for ecoregions of Africa and Madagascar. Different sets of priority ecoregions were identified when using different taxonomic groups and the non-species values did not correlate with species measures (Burgess et al. 2006). Marshall et al. (2006) however showed that a regionalisation built on macroinvertebrate data was supported by fish data.

The only sure way to assess biodiversity is to undertake comprehensive or complete inventories and to use this data to create aquatic regionalisations. However, as this is highly unlikely to occur due to resource constraints, methods are required that will provide some synoptic information relatively quickly and with validity, in conjunction with more comprehensive inventories in targeted systems. If biodiversity conservation is the primary goal, then in the short term this is likely to mean regionalisations based on surrogates, using one group of taxa to represent broader patterns of biodiversity. However, it is crucial to clearly state the purpose of the regionalisation and to show that the surrogates actually reflect broader patterns, not just assume they do.

### *3.3.1 Global frameworks – Dasmann-Urdvardy biogeographic framework and Olson et al. (2001) terrestrial ecoregions*

**Purpose:** Dasmann- Urdvardy: To create a classification of the world's biotic areas for conservation (Urdvardy 1975). Olson et al. (2001): Map terrestrial ecoregions of the world to identify areas of outstanding biodiversity and representative communities.

**Use:** Dasmann- Urdvardy: Global bioregionalisation developed for the World Conservation Union to guide establishment of representation in a global reserve network. Olson et al. (2001) ecoregions are intended primarily as units for conservation action.

**Key features:** Dasmann- Urdvardy: Qualitative; gives equal weight to structural and taxonomic differences of ecosystems (Jepson and Whittaker 2002); climatic and taxonomic variables are used. Olson et al. (2001): Qualitative; used existing biogeographic maps (including IBRA), landform and vegetation. Greatly increased resolution of the Dasmann-Urdvardy's framework, increasing the number from 198 (Dasmann) and 193 (Urdvardy) regions to 867 ecoregions.

Two global regionalisations are presented here, the Dasmann-Urdvardy system developed for the IUCN and the global ecoregion approach of WWF (Olson et al. 2001).

The Dasmann-Urdvardy regionalisation includes climatic variables; but the emphasis of the approach is focused on biological variables, flora and fauna. The top level of the regionalisation includes consideration of the biome system of Clements and Shelford (1939; cited in Jepson and Whittaker 2002) and Wallace's 1876 faunal regions. The second level is delineated by sub-dividing physiognomically defined climax vegetation type on the basis of faunal distinctiveness. The latter was assessed on areas with 65% species in common being considered separate faunal provinces. Fauna data was limited to bird and mammal data (Jepson and Whittaker 2002).

Olson et al. (2001) ecoregions are on average 150,000km2 (median 56,300km2), providing greater resolution than previous approaches. They produced 14 biomes and 8 biogeographic realms with 867 nested ecoregions. It is a predominantly biological regionalisation relying heavily on vegetation data as a surrogate for other taxonomic groups. Three caveats are made which relate are said to be relevant to all biogeographical mapping approaches (Olson et al. 2001):

1. No single biogeographic framework is optimal for all taxa;
2. Ecoregions reflect the best compromise for as many taxa as possible; and
3. Most ecoregions contain habitats that differ from their assigned biome.

Both of these regionalisations have global extents and as such are of limited value for application to identifying HCVAE at the national level. However, the terrestrial ecoregions identified for Australia were based on the IBRA bioregions.

### *3.3.2 Freshwater fish: Unmack (2001) - Australia*

**Purpose:** To investigate biogeographic patterns of obligate freshwater fishes in Australia.

**Use:** The results provide working hypotheses to be tested by phylogenetic analysis.

**Key features:** Quantitative; uses a combination of cluster, ordination and parsimony analysis to detect patterns.

Unmack (2001) used data from museums, the scientific literature, regional agencies and direct observation to provide presence absence data for 167 freshwater fish species at the river basin scale (largely following the AWRC drainage basins, but with some modifications). Relationships among regions (based primarily on taxon richness and endemism) were deduced largely by concordance between methodologies, then summarised into a proposed series of 10 provinces, several of which had subdivisions.

Tait et al. (2003) suggested that the regions produced by Unmack (2001) could be used as the first draft of the highest level of an Interim Freshwater Biogeographic Regionalisation for Australia (IFBRA), with the lower levels being based on drainage network position (Tait et al. 2003). However, Stein (2007) claims the Unmack provinces would require revision to reflect improved techniques for delineating drainage basins.

### *3.3.3 Freshwater fish: Growns and West (in review) - NSW*

**Purpose:** Trial a method for developing aquatic bioregional classification of rivers in NSW using predicted natural distributions of freshwater fish (Growns and West in press).

**Use:** Not explicitly stated, but suggested natural resource management and conservation planning.

**Key features:** Quantitative; six regions defined using non-hierarchical clustering. Based on “potential” natural species distribution, considered independent on actual fish sampling locations, and separation between zones is unambiguous. Spatial unit approximately one kilometer square grid cells.

Growns and West (in review) defined six regions in NSW using species predictive modeling. This was three more than Unmack (2001) defined for NSW, probably reflecting the different extent of the analyses udertaken. The use of potential natural distributions has its pros and cons. Although it is useful for targeted species conservation, Growns and West (in review) identify several potential limitations in the use of species predictive models relevant to conservation planning. False positives, where sites are identified as having the target species when the species is in fact absent, can lead to failure to conserve the species. False negatives could suggest a species is not present when in fact it is. Species distribution models can also have problems when using rare or uncommon taxa (Growns and West in review).

### 3.3.4 Macroinvertebrates bioregions Victoria

**Purpose:** Development of an *a posteriori* based on macroinvertebrate data to compare to IBRA, an *a priori* regionalisation (Marchant et al. 2000; Wells et al. 2002).

**Use:** Derivation of numeric biological objectives for the protection of river and streams in Victoria (Metzeling et al. 2002; Metzeling et al. 2006).

**Key features:** Qualitative; showed little correlation with IBRA regions found in Victoria. Final bioregions boundaries were delineated using a weight of evidence approach which included expert opinion.

Marchant et al. (2000) found that only 3 of the 10 IBRA regions had macroinvertebrate compositions that could be considered distinctive when compared to the other IBRA regions. The remaining IBRA zones, which cover approximately 60% of the State, showed relatively little compositional difference. Analysis of the 25 AWRC river basins in Victoria showed more correlation with the macroinvertebrate classification than did the IBRA regions (Marchant et al. 2000).

Wells et al. (2002) used a combination of classification and ordination techniques, expert opinion and environmental features to produce the regional boundaries of the bioregions, which were then used to develop aquatic macroinvertebrate-based river protection objectives. Although defined using macroinvertebrate distributions *a posteriori*, the five regions identified strongly reflect altitudinal and associated physiographic features, which are key factors in *a priori* regionalisations (Metzeling et al. 2006). The regionalisation provided the framework for setting biological objectives specifically based on reference site conditions within each separate region. Ultimately this led to the revision of the State of the Environment Protection Policy (Waters of Victoria), which aims to provide specific environmental quality management objectives for the State of Victoria (Wells et al. 2002; Leon Metzeling, Victorian EPA, pers. comm.). The five regions produced are not considered to be relevant to establishing HCVAE as they were developed for a separate reason and focus on one aspect of aquatic biodiversity, namely macroinvertebrates.

### 3.3.5 Freshwater Bioregionalisation of QLD riverine ecosystems

**Purpose:** Develop a regionalisation that balances the need for a spatial framework against realistic monitoring and assessment program requirements.

**Use:** Water resource management – monitoring and assessment planning framework.

**Key features:** Qualitative; described as a bottom-up approach using macroinvertebrate and fish data with ecosystem drivers considered secondarily. Management requirements predetermined the maximum number of regions to be developed and expert opinion was used to help delineate the final boundaries.

An *a posteriori* approach was used to identify geographic areas with similar faunal assemblages. Macroinvertebrate data collected using the AusRivAS rapid assessment protocol (reference sites only) was used to produce the regionalisation. As the initial cluster analysis based on the approach used in Victoria, showed only a separation of the coastal area of north QLD ("wet tropics") from the rest of the state, catchment boundaries were imposed as the basis of the regionalisation. Where catchments had insufficient data, expert opinion was used to help delineate regional boundaries (Marshal et al. 2006). Nine regions were identified, referred to as Freshwater Biogeographical Provinces, using drainage basin boundaries as the 'starting point'. Management requirements restricted the number of regions identified to less than 10 in order to maintain a certain frequency of assessment within existing resource constraints (Department of Natural Resources, Mines and Water, 2006; Marshall et al. 2006). Also regions were required to be contiguous single entities rather than disjointed patches of the same region type over the landscape (QLD) (Marshal et al. 2006). Fish data, excluding species with strong marine or estuarine affinities, were used to establish the convergence of the regions for other biota, with the fish data supporting the regionalisation defined by the macroinvertebrate data and expert opinion (Marshall et al. 2006). As with other bottom up approaches reviewed here, there are limitations for application at the national extent due to data limitations.

## 3.4 Ecosystem driver based regionalisations

As discussed in section 2.2.2 drivers or controlling factors are the abiotic variables which strongly influence, or drive, environmental responses. Mackey et al. (2008) describe ecosystem drivers as mainly exogenous physical environmental variables that can approximate the primary environmental regimes. Drivers can include climatic and atmospheric parameters, substrate attributes (soils/regolith), terrain attributes, and data that describe natural and human induced disturbances regimes (fire, flooding, land clearing/land use changes) (Mackey et al. 2008). For aquatic ecosystems one of the principal drivers is hydrology, with hydrological connectivity considered critical to the functioning and integrity of aquatic ecosystems. Surface hydrological units could be considered a terrain attribute, but are emphasized here as a key variable for consideration in this type of regionalisation for aquatic ecosystems.

Many approaches assigned to this type of regionalisation also include terrestrial vegetation formation considerations and thus could be considered as biological regionalisation types – as indicated above, our groupings of regionalisations is subjective.

### *3.4.1 Environmental domains: Mackey et al. (2008) - Australia*

**Purpose:** To test a generic framework designed to facilitate the systematic approach to developing biogeographical regionalisations. The assumed purpose for the case study is to provide national level information relevant to planning conservation of biodiversity and related natural heritage values.

**Use:** The regionalisation was designed to test the main thematic foci of “ecosystem drivers” of the proposed framework, by creating a classification of environmental domains using 11 climatic, terrain and soil attributes. Mackey et al (2008) suggest this type of regionalisation could be used to address questions relating to the relationship between environmental distance and evolutionary processes.

**Key features:** Quantitative; proposes a generic framework for developing biogeographic regionalisations. Uses surface hydrological units as the spatial unit of analysis mapped at 1:250,000.

Mackey et al. (2008) detail a generic framework (i.e. relevant to all terrestrial landscapes) for developing biogeographic regionalisations. They tested the framework by producing two regionalisations, one based on ecosystem drivers (the environmental domain regionalisation) and one based on ecosystem responses (the primary productivity regime regionalisation - see below for discussion on the primary productivity regionalisation). Surface hydrological units were used as the spatial framework as they relate to local fluxes of water and nutrients and vary in size according to terrain complexity. Where stream sections were not mapped, low lying depressions and ephemeral lakes were included using digital elevation models. Overall, approximately 1 million surface hydrological units formed the basis of the regionalisations. Mackey et al. (2008) adopted a numeric parametric approach and outputs of the regionalisation included a climatic classification, which generated 30 groups.

The environmental domain regionalisation and climate classification were based on the variables shown in Table 9. The environmental domain regionalisation produced 151 groups. Notable features of the regionalisation were a finer level of patterning compared with the climate classification due to the inclusion of the terrain and soil attributes. Major floodplain regions were identified for the Murray River and also the inland channel country. Geographically large or extensive groups typically have few topographic features and are characterised by their surface hydrological units (or lack of surface water in some instances) and distinctive substrates. The variables used to produce the regionalisation reflect natural processes and as such show potential rather than actual ecological patterns. Current condition or degree of human modification is not included in the regionalisation. The productivity regionalisation does include consideration of human impacts on land cover (see below) (Mackey et al. 2008).

Comparisons with IBRA regions were made for each regionalisation. As each State and Territory identified its own IBRA regions, there were effectively 7 different approaches and a lack of consistency in the methods used for delineating regions. This, and the fact that IBRA includes consideration of patterns of taxonomic composition and evolutionary legacy that characterise the regions, means that the regionalisations produced by Mackey et al. (2008) cannot be considered as replacements for IBRA.

**Table 9: Variables used as attributes in the climatic and environmental domain regionalisations (Mackey et al. 2008).**

| Variable | Spatial statistic calculated for each SHU |
|---|---|
| Annual mean temperature (˚C) | Mean |
| Maximum temperature of warmest month (˚C) | Mean |
| Minimum temperature of coldest month (˚C) | Mean |
| Precipitation of coldest quarter (consecutive 3 months; mm) | Mean |
| Precipitation seasonality (coefficient of variation based on long term monthly means) | Mean |
| *W*, annual availability of water (mm) | Mean |
| Catchment contributing area ($km^2$) | Mean and maximum (log) |
| Surface slope (˚) | Mean and maximum (log) |
| Flatness class (categorical data)<br>0 Erosional<br>1 Indeterminate<br>2 Valley bottom flats<br>3 Ridge top flats | Mode |
| Surface drainage class (categorical data)<br>1 Direct external<br>2 Indirect external<br>3 Coordinated interior<br>4 Uncoordinated interior<br>5 Riverless | Mode |
| Substrate class (categorical data)(not shown here) | Mode |

### *3.4.2 Drainage basins: Australian Water Resource Council*

**Purpose:** The AWRC drainage system was developed for States and Territories to manage and report on surface and groundwater.

**Use:** Water management.

**Key features:** Assumed quantitative but method was not sourced. The drainage divisions and basins/catchments are suggested by Kingsford et al. (2005) as a reasonable spatial unit for assessment and management of HCVAE. The data is available at three scales for the continent (Kingsford et al. 2005). Drainage divisions and basins/catchments do not represent a complete regionalisation method, only the spatial unit. Decisions regarding the variables used as attributes to develop an aquatic ecosystem regionalisation would still need to be made.

Drainage basins have been included under the ecosystem driver category of regionalisation, as they capture the key drivers of aquatic ecosystems – hydrology and hydrological connectivity. They do not represent a regionalisation as such, but rather the spatial unit on which a regionalisation can be based.

*Australian Water Resources 2005* has mapped the management boundaries used by the States and Territories to manage and report on surface and groundwater. Surface water is divided into:

- 12 drainage divisions (see Figure 10);
- 246 river basins; and
- 340 surface water management areas (SWMAs).

Groundwater is divided into:

- 69 groundwater provinces; and
- 367 groundwater management units (GMUs).

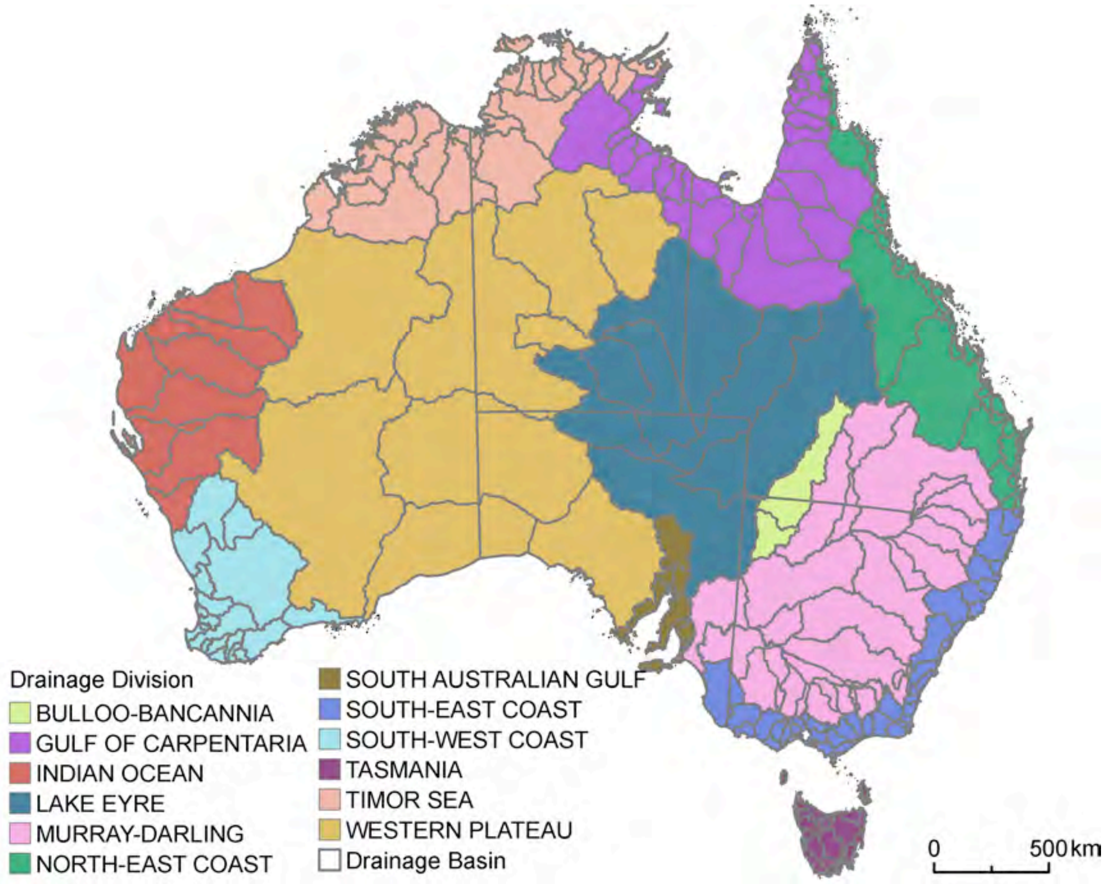


**Figure 10: AWRC surface water drainage system. Data source AUSLIG 1997 (from Stein 2007 reproduced with permission).**

Kingsford et al. (2005) called for a national spatial framework for the identification and management of HCVAE and recommended the use of the Australian Water Resource Commission (AWRC) drainage basin framework. Initially, catchments were suggested as the basic spatial unit for assessment and management, but Kingsford et al. (2005) acknowledged that as delineation of this scale was relatively recent for the continent, that drainage basins were a more appropriate unit. This does not represent an actual method of regionalisation, only a recommendation regarding a spatial unit. Kingsford et al. (2005) consider the current river basin network as being an adequate spatial framework, which captures the main issue of hydrological connectivity. The spatial hierarchy that Kingsford et al. (2005) suggest is presented in Table 10, with drainage divisions, river basins/catchments and river segments as being the main units for which data is currently available at the continental scale.

**Table 10: Suggested hierarchy of spatial units comprising a drainage division from Kingsford et al. (2005).**

| Scale | Description | Time scale of continuous potential persistence (years) | Linear spatial scale (stream length) | | Applicability at continental scale |
|---|---|---|---|---|---|
| | | | Small streams (m) | Large streams (km) | |
| Micro-habitat | Patch of similar flow velocity, substrate cover | $10^1$-$10^0$ | 0.1 | 0.1 | Not possible |
| Habitat/bed form | Areas of relatively homogeneous bed material, flow velocity and depth | $10^0$-$10^1$ | 0.1-10 | 0.1-10 | Not possible |
| Reach | Length of river exhibiting relatively homogeneous channel characteristics or a consistent pattern of repetitive/alternating characteristics | $10^1$-$10^2$ | 10-100 | 10-100 | Not possible currently, prohibitive resource requirements |
| Segment (link) | Portion of stream and its floodplain (including associated wetlands), bounded by tributary junctions, major waterfalls or lakes. The area of land draining to a segment or groups of segments is a sub-catchment. | $10^3$-$10^4$ | 100-1000 | 100-1000 | Currently possible |
| Catchment | The area of land drained by a stream to a particular point (e.g. a tributary junction). May include internal sub-catchments | $10^4$-$10^5$ | >1000 | >1000 | Currently possible |
| River basin | All of the catchment area that drains to a river mouth or terminal lake | $10^5$-$10^6$ | 1-100km | 1000-10,000 | Currently possible |
| Drainage division | Grouping of river basins according to discharge point, geography and or climate | $10^5$-$10^6$ | na | >10,000 | Currently possible |

The use of drainage basins alone is river centric, as it does not include consideration of karst, groundwater dependent or non-floodplain wetlands. Mangroves and saltmarsh ecosystems are not explicitly accounted for in this system, as segments are suggested as linear features only (see Figure 11).

Kingsford et al. (2005) made the following recommendations with regard to developing a spatial framework for identifying HCVAE:

- Use current drainage divisions, river basins and river segments for initial implementation of this framework. These map layers, and the sub- catchments and catchments they support, should be publicly available.
- River ecosystem data should be labelled according to resolvable hierarchical scales allowing for future evaluation and reassessment of classifications.
- Develop a new hierarchical spatial framework for managing aquatic systems and rivers, based on topography and drainage networks and without the problems of current spatial layers.

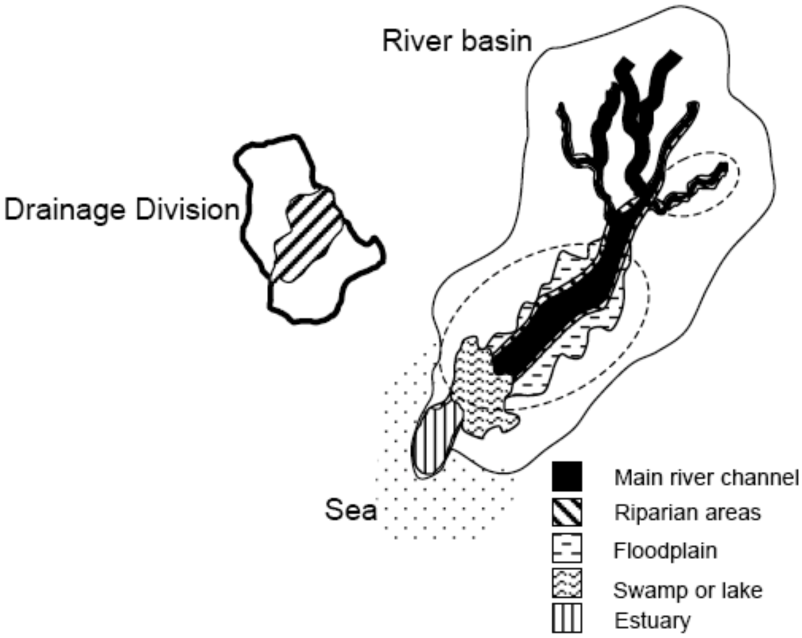


**Figure 11: Diagram of theoretical river basin within a drainage division (inset), showing dependent ecosystems (main river channel, riparian areas, floodplains, swamps or lakes and an estuary). Dotted lines indicate potential river segments within this river basin (from Kingsford et al. 2005).**

### *3.4.3 Landscape framework for systematic conservation planning for Australian rivers and streams: Stein (2007) - Australia*

**Purpose:** Develop and test a continental spatial framework applicable to Australian river systems. Develop and evaluate a landscape classification for Australian rivers.

**Use:** Conservation planning.

**Key features:** Quantitative; nested hierarchical spatial framework developed in line with the NRS guidelines. Suitable for use at the regional and continental scales.

Stein (2005, 2007) described a continental nested hierarchical catchment framework on which riverine conservation planning can be based. The scale of the drainage analysis (approximately 1:250,000) is suitable for planning at multiple scales (regional to national) and has been developed to be consistent with the scientific guidelines of the National Reserve System (Stein, 2005; 2007). The framework provides an alternative delineation of surface hydrological units to the AWRC drainage divisions. Stream networks and drainage basins are delineated and subdivided using the Pfafstetter scheme, which defines the main stem as draining the larger portion of the catchment. Surface flow paths derived from the national 9 second digital elevation model (Hutchinson et al. 2001; Geoscience Australia 2001a) were used to delineate catchments (Figure 12).

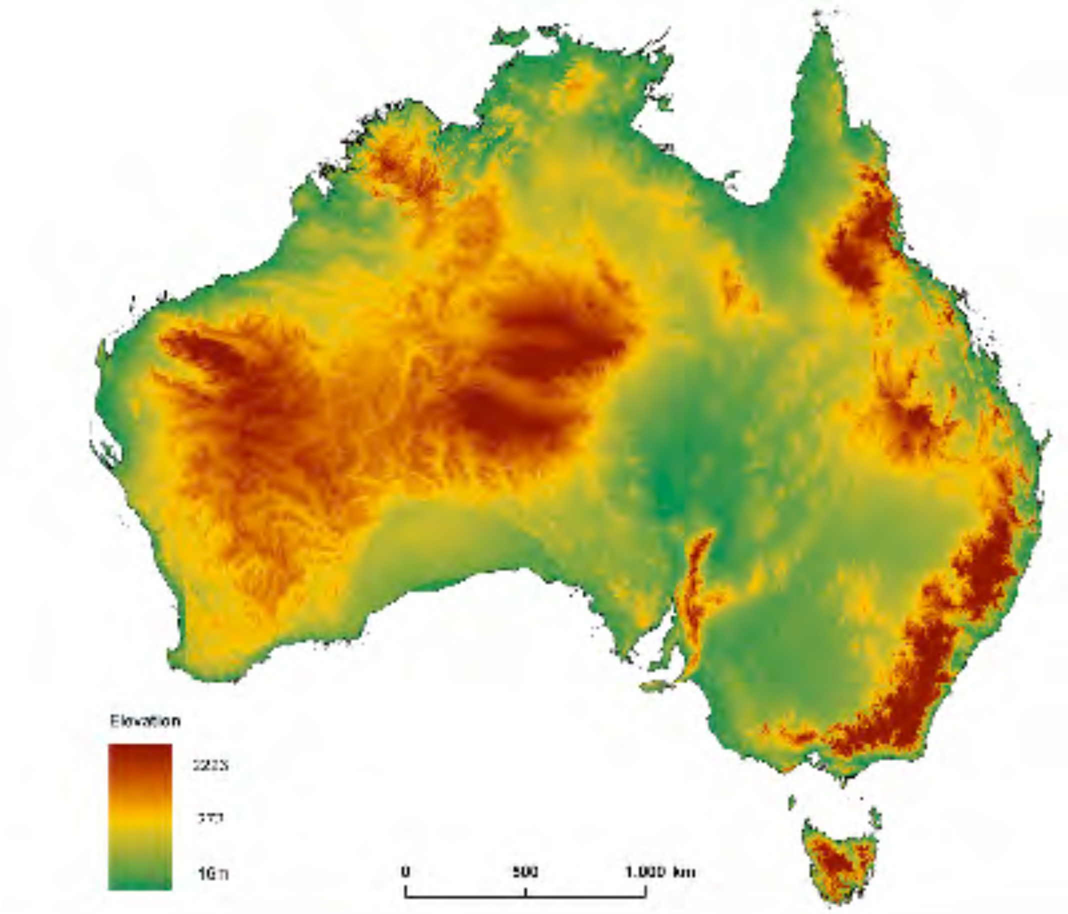


**Figure 12: The 9 second DEM version 2.1. Data source Hutchinson et al. (2001) and Geoscience Australia (2001a) (from Stein (2007) reproduced with permission).**

Stein (2007) also produced 6 landscape scale classifications of stream networks based on attributes relating to climate (Figure 13), geology, catchment water balance (referred to as flow classification), terrain, vegetation (pre European, not current) and a combination classification called ALL (Figure 14). The number of end groups was limited to 10 for each classification, which were then combined to produce 5 hierarchical classification levels with climate considered the single controlling factor at the highest level. Hierarchical classification levels ("Levels 1 to 5") were formed by combining the 10 group classifications as follows (Stein 2007):

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

River Environment Types were defined by the combination of a stream segment's group memberships (Stein 2007). The environmental domain approach that produced the single classification (ALL) produced relatively heterogeneous groupings due to scale differences in the attributes. The environmental characteristics of the River Environment Types formed by combination of separate landscape classifications were more distinctive. Whilst the hierarchical classification was considered suitable to conservation planning at the continental scale, it defined a much larger number of River Environment Types than ALL.

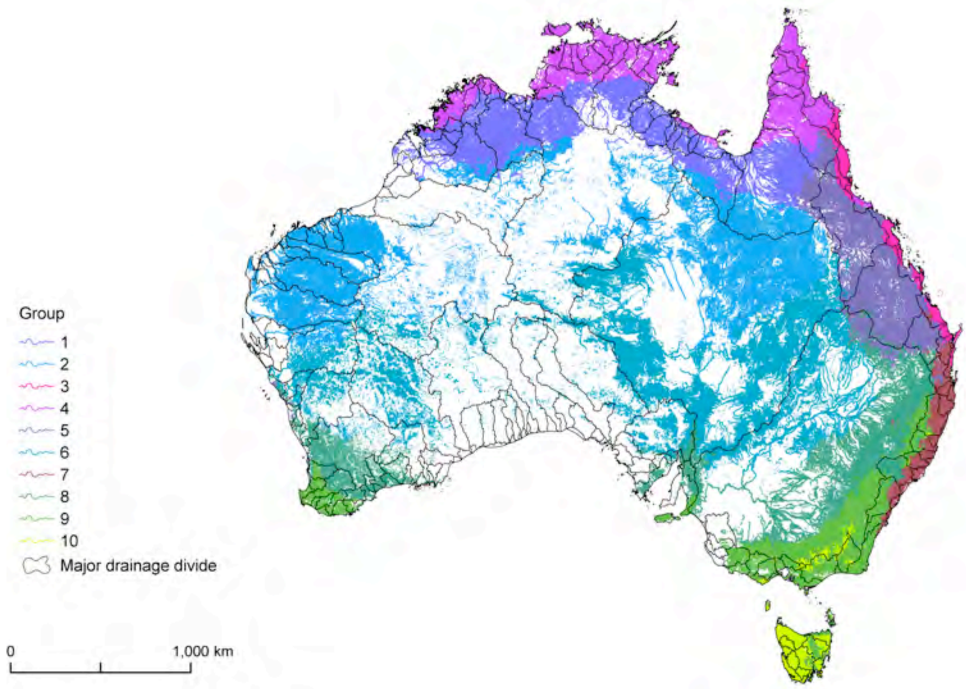


**Figure 13: Stein (2007) classification of stream segments based on climatic attributes (from Stein (2007) reproduced with permission).**

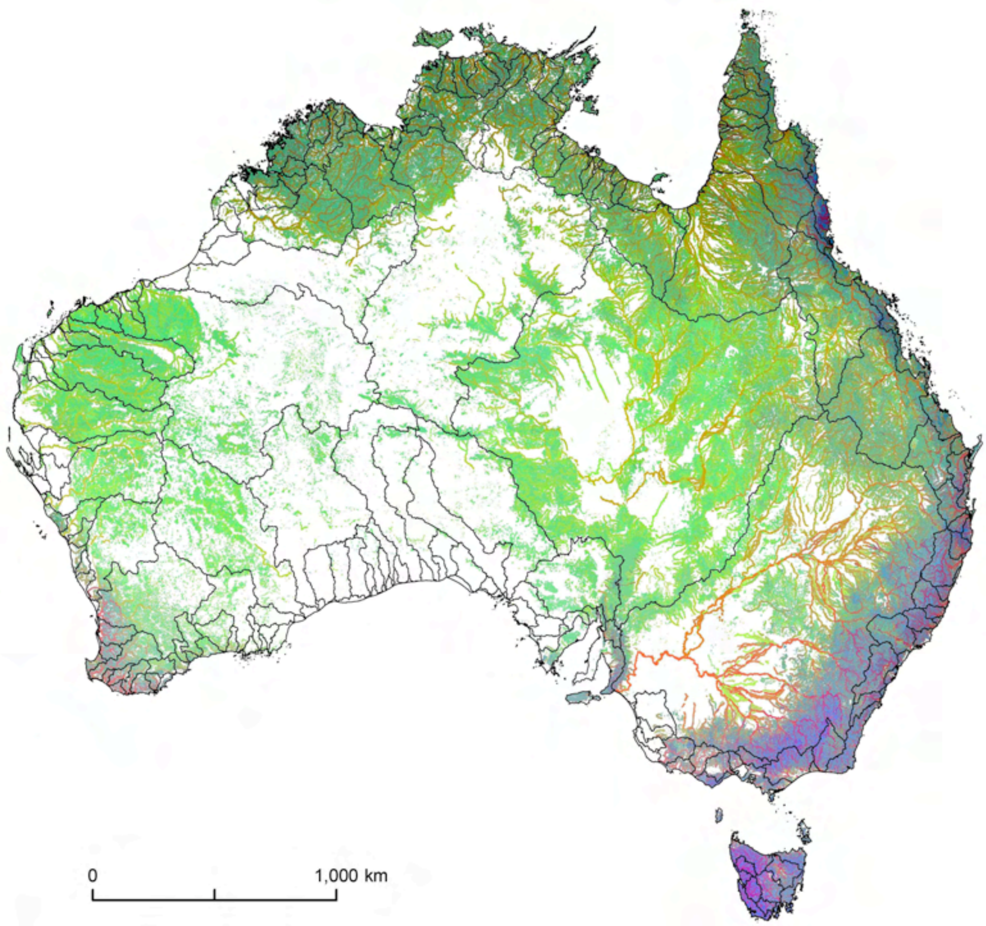


**Figure 14: Stein (2007) ALL classification – legend of 355 groups/colours not reproduced here (from Stein (2007) reproduced with permission).**

The framework provided by Stein (2007) addresses issues/limitations associated with other drainage schemes, such as the AWRC drainage system, as the catchments boundaries are defined strictly on a topographic basis. The stream network delineates the extent of the channelised flow and the natural variation in drainage density at the map scale of 1:125,000. The stream segments, which are used in the landscape-scale classifications, represent the finest scale of the spatial framework and are suitable units for further analysis once additional data becomes available (e.g. biotic layers) (Stein 2007). The framework describes potential ecosystems, not current conditions, which aligns with the target setting principles of the guidelines of the NRS. Compliance with the NRS guidelines are shown in Table 11. The framework was used to test the comprehensiveness and adequacy of the NRS with regard to river ecosystems, highlighting its applicability at the continental scale for conservation planning. It represents one of the most comprehensive and relevant of the regionalisation frameworks reviewed, however it does not include consideration of estuaries. Aquatic ecosystems included were river and streams and ecosystems dependent on river flows including complex effluent channels, terminal and floodplain wetlands (Stein, 2007).

**Table 11: Compliance with the target setting principles established for the National Reserve System (from Stein 2007).**

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

### *3.4.4 River Environmental Domain Analysis: Jerie et al. (2003) – Tasmania*

**Purpose:** To produce a geomorphic classification of rivers at the landscape scale and a regionalisation of river types.

**Use:** Water resource management including a context for conservation planning.

**Key features:** Quantitative: produced a river classification based on geomorphological attributes, and environmental domain analysis to identify breaks in the system controls (drivers) to produce a spatially hierarchical regionalisation at three scales. Requires ground truthing and ongoing refinement.

The primary attributes used in the Tasmanian River Environmental Domain Analysis (EDA) are considered system controls (drivers) of river development and behaviour. Attributes include topography, geology, climate, and history of geomorphic processes. The spatial scale of analysis is a 200 m grid of Tasmania. Thousands of patches were produced and classified into 489 river environmental domains (Jerie and Houshold 2003). At the landscape scale, analysis of the domains identified 90 (Ian Houshold, DPIW, pers. comm.) areas with repeating features, called sub regional mosaics, or fluvial landscapes. These sub regions occupy 10's of square kilometers. The intention is to use the distribution pattern of the sub regions and how rivers flow through particular sequences of sub regions to identify river regions for the state (Jierie et al. 2003). The method is based solely on geomorphology and does not include any biological, hydrological, or water quality variables, all of which would be required to produce a bio-physical regionalisation. The final regionalisation has yet to be completed.

### *3.4.5 Ecoregion frameworks: Bailey-Omernik - USA*

**Purpose:** Bailey's ecoregions were originally developed for ecosystem management and Omernik's for water quality assessment.

**Use:** Wide application and modification of the basic frameworks has occurred in the USA, among other places.

**Key features:** Qualitative. Bailey used an approach that assumes a single environmental factor associated with ecological processes acts as the primary control for regionalisation at a given scale – a factor based classification. This is referred to as an environmental controlling system approach by McMahon et al. (2004). Omernik and others used an environmental synthesis approach or weight of evidence approach which considers ecological regions the net result of the interplay of biophysical components, the relative importance of each varying between locations (McMahon et al 2004).

A number of ecoregion approaches have been developed in the United States (Table 12). The most well know are those of Bailey (1987), Omernik (1987) and the Nature Conservancy (TNC). The WWF US ecoregions approach is a modification of the Bailey and Omernik approaches, as is the Common Ecological Regions (CER) approach applied to the United States (McMahon et al. 2001). The TNC and WWF US methods are considered in more detail in sections 3.4.6 and 3.4.7.

The main use of ecoregions in the United States is for running environmental resource regulatory programs and the development of management strategies (Omernik and Bailey 1997). Comparisons of ecoregions with regionalisations based on fish assemblages, selected hydrological characteristics, macroinvertebrate distributions and wildlife communities have shown correlations however there were also other spatial characteristics that helped explain relationships (see references within Omernik and Bailey 1997).

As with the regionalisations considered previously, the ecoregion maps derived from Bailey-Omernik framework share common roots but require iterative improvements and modifications for use in different applications. Omernik's ecoregions have four levels and is implemented hierarchically, having successively finer spatial and categorical resolution. Bryce and Clarke (1996) extended Omernik's framework to a more detailed level to bridge the gap between stream habitat and State-level regional classifications.

Loveland and Merchant (2004) state that both the Bailey and Omernik methods have been criticized for the decision logic and methods employed. Bailey's contention that climate is the single controlling factor can be disputed and, as many controlling factor(s) vary aerially, this need to be taken into consideration otherwise the end result is a single theme map rather than an integrated ecoregion map (Bryce and Clarke 1996). Fortunately, the two approaches to ecoregion mapping share important characteristics, and their similarities probably outweigh differences (Bryce and Clarke 1996); as each addresses the needs of a particular audience, there is room for both (and other) methods (Loveland and Merchant 2004).

**Table 12: Major ecoregion approaches and their applications for the United States (modified from Loveland and Merchant 2004).**

| System | Sponsor | Derivation | Intended application |
|---|---|---|---|
| Omernik ecoregions | US Environment Protection Agency | | Water quality assessment |
| Bailey ecoregions | US Forest Service | | Ecosystem management |
| Major land resource (MLR) areas | Natural resources conservation service | | Agricultural reporting and conservation programs |
| Common ecological regions (CER) | National Interagency Technical Team (consortium) | Derived from Bailey, Omernik ecoregions and MRL | Ecosystem management and interagency coordination |
| Committee for Environmental cooperation | North American Free Trade Agreement Committee for Environmental Cooperation | Modified Omernik and maps from Canada and Mexico | State of the environment reporting |
| Bird Conservation Areas | North American Bird Conservation Initiative | | Habitat conservation and management |
| TNC ecoregions | The Nature Conservancy | Modified from Bailey's | Conservation planning, species protection |
| WWF ecoregions | WWF US | Modified from Omernik's | |

### *3.4.6 TNC freshwater classification for ecoregional assessment: Higgins et al. (2005) – North America and elsewhere*

**Purpose:** To generate coarse filter conservation targets, with targets being elements of biodiversity that are a focus for conservation planning (Higgins et al. 2005).

**Use:** Freshwater biodiversity conservation planning.

**Key features:** Qualitative. Considers conservation of whole ecosystems to move beyond species based conservation. Methodology used to implement the classification is based on practical needs, being considered a rapid approach which is compatible with readily available data. Can be used in different locations; describes a spatial framework with suggested variables that can be used to define the four hierarchical levels.

The TNC ecoregion approach to freshwater conservation is considered a ecosystem driver regionalisation, as the primary variables used are abiotic at the finer levels of the approach. However, as biological data (typically fish distribution data) are used at the higher planning levels, this approach could also be considered a taxonomic composition and evolutionary legacy regionalisation.

The freshwater classification is used to support ecoregion assessments and generate coarse filter conservation targets in a systematic way. The classification was specifically developed with the aim of identifying elements of aquatic biodiversity that are the focus for conservation efforts, and to be applicable at the ecoregion scale (Higgins et al. 2005). It is a four-tiered hierarchical system, based on spatial data at scales known to drive freshwater biodiversity patterns:

1. **Aquatic zoogeographic units:** also called Freshwater ecoregions by Silk and Ciruna (2004); scale approx 1:26,000,000.Typically conforms to drainage boundaries, but not always as watersheds may need to be subdivided in certain circumstances. They are distinguished by patterns in zoogeography, often patterns of fish distributions. Where available these can be adopted from other sources. This level represents the overall planning unit.

2. **Ecological drainage units:** nested within one aquatic zoogeographic unit; scale approximately 1:26,000,000. Represents a finer scale of physiographic and zoogeographic diversity that allows the selection of rivers and wetlands for conservation to be stratified. Qualitative interpretation of biotic data from available sources is used to delineate boundaries. They typically conform to patterns of physiography, climate and freshwater ecosystem connectivity. Ecoregion maps (e.g. WWF, Omernik, Bailey for North America) can be used as a source of spatial data.
3. **Aquatic ecological systems;** called meso-scale units by Silk and Ciruna (2004); and nested within ecological drainage units; scale approximately 1:4,000,000. These represent coarse filter targets (equate to HCVAE) and are delineated using abiotic variables. Sizes of aquatic ecological systems range from 100 $km^2$ headwater streams, lake and wetland complexes to the largest riverine catchment in an ecoregion.
4. **Macrohabitats:** called local scale units by Silk and Ciruna (2004) and nested within aquatic ecological systems; scale approximately 1:1,200,000. These are typically only generated when there is sufficient hydrological data available to facilitate an automated processing of data. They are delineated using abiotic variables.

Higgins et al. (2005) provide a list of standard abiotic variables used in delineating their aquatic ecological systems and macrohabitats, along with two case studies illustrating the application of the method.

### *3.4.7 WWF freshwater ecoregions: Abell et al. (2002) – North America and elsewhere*

**Purpose:** Conservation of freshwater biodiversity.

**Use:** Freshwater biodiversity conservation assessment and planning.

**Key features:** Qualitative; uses a catchment framework to describe freshwater (inland) hydrologically connected (to rivers) ecoregions that can be applied at multiple scales. Recommend starting at the coarser scale and sub-dividing as data limitations at the finer scale often prevent starting with smaller sub-catchments and combining them into progressively larger units.

Abell et al. (2002) use the term freshwater to include all inland waters, whether fresh or saline, in their freshwater ecoregional approach. The focus is predominantly on aquatic ecosystems that are hydrologically connected, stating that wetlands that are not hydrologically connected to rivers, lakes or springs are adequately covered in terrestrial ecoregion approaches to conservation planning. Abell et al. (2002) acknowledge that geoclimatic features such as elevation, relief, slope, landform, soil and drainage density largely drive freshwater systems, but that terrestrial vegetation patterns do not necessarily reflect aquatic biotic patterns. Obligate freshwater taxa, such as freshwater fish, are suggested as the most appropriate basis for delineating freshwater ecoregions. However, because of limited data on freshwater fish distributions, they adopt catchments as a "logical starting point" for delineating freshwater ecoregions.

Prioritising freshwater ecoregions for conservation planning is based on consideration of areas of biological importance related to species richness, endemism, evolutionary phenomenon, and other attributes; the level of ecological integrity and risk of future threats also contribute to prioritising systems for conservation investment (Abell et al. 2002).

WWF has completed continental conservation assessments of freshwater ecoregions for North America, Latin America and the Caribbean, Africa, Madagascar and Indonesia (Olson et al. 1997; Abell, et al. 2002; Wikramanayake et al. 2002; Thieme et al. 2005). The assessments constitute work aimed at mapping freshwater ecoregions for the world detailing biodiversity and threat information for the units. Effectively this leads to a prioritisation of freshwater ecoregions (Phillips and Butcher 2006). There has been some minor criticism of WWF terrestrial ecoregion approach in the literature, with Jepson and Whittaker (2002) arguing that the method failed to significantly advance the use of ecoregions within Indonesia, a claim refuted by Wikramanayake et al. (2002).

## 3.5 Ecosystem response based regionalisations

Mackey et al. (2008) states that regionalisation based on ecosystems response variables enables exploration of current conditions and impacts. The primary variables on which regionalisations could be based include vegetation productivity, vegetation structure and condition. Structure refers to height, density and layering of vegetation cover and condition relates to the extent to which vegetation has been altered or degraded by anthropogenic impacts (Mackey et al. 2008). There are relatively few existing regionalisations that fall into this category that do not also include driver variables, as detailed in the previous section.

### *3.5.1 Primary productivity regimes: Mackey et al. (2008) - Australia*

**Purpose:** To test a generic framework designed to facilitate the systematic approach to developing biogeographical regionalisations. The assumed purpose for the case study is to provide national level information relevant to planning conservation of biodiversity and related natural heritage values.

**Use:** Mackey et al (2008) suggest this type of regionalisation could be used to address questions relating to productivity regimes and animal life history strategies.

**Key features:** Quantitative; proposes a generic framework for developing biogeographic regionalisations. Uses surface hydrological units as the spatial unit of analysis, mapped at 1:250,000 with segments equating to the section of stream between tributary junctions. Mackey et al. (2008) chose to adopt an analytical model using a numeric agglomerative approach, resulting in complex spatial patterns with groups having more than one geographical occurrence with numerous outliers.

Mackey et al (2008) detail a generic framework (i.e. relevant to all terrestrial landscapes) for developing biogeographic regionalisations. They tested the framework by producing two regionalisations, one based on ecosystem drivers – the environmental domain regionalisation (see above) and one based on ecosystem responses – the primary productivity regime regionalisation. Surface hydrological units were used as the spatial framework because it was argued they relate to local fluxes of water and nutrients and vary in size according to terrain complexity. Where stream sections were not mapped, low-lying depressions and ephemeral lakes were included using DEM. Overall, approximately 1 million surface hydrological units formed the basis of the regionalisations. Remotely sensed data of current land cover were used in the analysis and as such the patterns shown in the regionalisation are a composite of natural biogeographic patterns and those caused by human changes to land cover (Mackey et al. 2008).

## 3.6 Holistic regionalisations

Holistic regionalisation approaches are those that consider abiotic drivers (climate, terrain, geomorphology, soils and hydrology) as well as biotic components. Most of the regionalisations reviewed give some consideration to abiotic and biotic attributes, however most of the holistic approaches would be considered top-down approaches where the biotic data is typically used to sub divide regions.

### *3.6.1 Interim Biogeographic Regionalisation for Australia (IBRA)*

**Purpose:** Scientific framework for the National Reserve System.

**Use:** State of the Environment reporting, conservation planning, in particular identifying gaps in the NRS with regards to achieving comprehensive cover of ecosystems at the national scale.

**Key features:** Qualitative; weight of evidence approach, relying on interpretation and synthesis of best available datasets by each State and Territory to produce regions. Consideration of aquatic ecosystems was not explicit. Wide acceptance for use in State of the Environment reporting.

The Interim Biogeographic Regionalisation for Australia (IBRA) was developed in mid 1990s as the scientific framework for the National Reserve System (NRS). Thackway and Cresswell (1995) documented the initial development phase of IBRA in which 130 biogeographic regions were refined to a total of 80 IBRA regions across Australia. Since then, a number of iterative refinements has resulted in Australia being divided into 85 regions and 403 sub-regions. The method uses an integrated survey approach, in which information on regions (provided by each State) were refined by expert judgment. The majority of the data used focused on vegetation communities and land system mapping.

The main attributes used in the development of IBRA were (Thackway and Cresswell 1995):

- Climate,
- Lithology/Geology,
- Landform,
- Vegetation,
- Flora and Fauna,
- Land use, and
- Other attributes if needed.

The relative contribution of the attributes used to delineate IBRA boundaries and to describe the IBRA regions varied by jurisdiction due to differences in the scale of datasets available (Table 13).

Thackway and Cresswell (1995) recognised the likelihood of IBRA being used for applications other than its original purpose and warned against this, suggesting caution needed *"to be taken in the application of regionalisations for wild rivers, old growth forests, wilderness, rare and endangered species and communities and National Estate values".* IBRA does not account for catchment constraints that influence obligate freshwater species, such as fish, nor the connected nature of aquatic ecosystems (Unmack 2001; Stein 2007).

Tait et al. (2003) suggest an *Interim Freshwater Biogeographic Regionalisation of Australia (IFBRA)* could be based on the natural biogeographic units provided by river basins, which would form the macro-regions of a hierarchical framework with the second scale of the hierarchy defined by sub-drainage regions. Primary attributes suggested for use in the IFBRA include distribution data for biota, phylogenies, and molecular data. Different aquatic biota have different dispersal capabilities and as such the vagility of different components of aquatic biodiversity should be recognised a key determinant of any aquatic bioregional framework (Tait et al. 2003). Tait et al. (2003) acknowledged that for some elements of aquatic biodiversity, terrestrial based regionalisations such as IBRA would be an appropriate planning framework. Patterns of biodiversity in surface non-riverine wetlands are likely to reflect both terrestrial and aquatic derived bioregions. Groundwater dependent systems and karsts may reflect some degree of bioregional distinction between drainage basins (Tait et al. 2003).

**Table 13: Indicative list data sets used in developing IBRA (from Thackway and Cresswell 1995)**

| State/ Territory | Data sets used |
|---|---|
| ACT | Vegetation type data - 1:25 000 |
| NSW & ACT | Coarse scale (1:100,000) vegetation mapping for eastern one-third of NSW derived from Landsat TM imagery (NSW NPWS) |
| | Detailed vegetation maps (1:250,000) for selected areas of NSW (Royal Botanic Gardens) |
| | Land systems in western NSW (NSW CaLM) |
| | Thematic soils maps for parts of NSW (NSW CaLM). |
| | Riparian vegetation mapping for selected major rivers in NSW (NSW Dept. of Water Resources) |
| | Natural Regions of NSW, incorporating Morgan and Terrey (1992) |
| | Regional and provincial mapping at 1:1,000,000 scale |
| | Geology mapping (BMR and NSW Dept. of Mineral Resources) |
| | Vegetation model for SE NSW (NSW NPWS) |
| | Vegetation maps for NE NSW (NSW NPWS and State Forests of NSW) |
| | State Environmental Planning Policy 26 - Littoral Rainforest (NSW Dept. of Planning) |
| | State Environmental Planning Policy 14 - Coastal Wetlands (NSW Dept. of Planning) |
| | Digital Elevation model of NSW (NSW NPWS) |
| | Climatic surface models of NSW (NSW NPWS) |
| | Predictive models of species distributions (Course scale for the whole of the State, detailed models for selected regions) (NSW NPWS) |
| NT | Land system mapping developed by CSIRO and CCNT |
| | Vegetation mapping |
| | Environmental domains |
| | Biogeographic domains |
| QLD | Land system mapping at 1:500,000 scale |
| | Biogeographic Regions at 1:2,500,000 scale |
| SA | Environments of South Australia (Laut et al 1977) 1:500,000 scale |
| TAS | Nature Conservation Regions (Orchard 1988) 1:500,000 scale |
| WA | Beard (1980) |
| | Geological Map of Western Australia, 1: 250,000. Geol. Survey of WA. (Myers, J.S. and Hocking, R.M. (compilers), 1988.) |
| VIC | Land systems and geomorphic units developed by the Land Conservation Council at 1:500,000 scale |
| | Flora of Victoria (1993) |
| Continental | Vegetation units derived from NOAA satellite data (ERIN 1991) |
| | Geology (BMR series at 1:2,500,000) |
| | Vegetation of Australia (AUSLIG 1990) 1:5,000,000 |
| | Digital Soil Atlas of Australia (NRIC and CSIRO 1991) |

### 3.6.2 Interim Marine and Coastal Regionalisation of Australia (IMCRA)

**Purpose:** National and regional framework for Australia's National Representative Systems of Marine Protected Areas (NRSIMPA).

**Use:** Marine planning and management.

**Key features:** Qualitative; explicitly acknowledges use of both qualitative and quantitative methods; however it is defined here as an integrated survey approach that uses best available data.

The Interim Marine and Coastal Regionalisation of Australia (IMCRA v3.3) has been combined with the National Marine Bioregionalisation (NMB) for off-shelf waters (Commonwealth of Australia 2006) to create IMCRA v.4.0. IMCRA is hierarchical in nature and recognises ecological patterns and processes at the continental, regional, local and site scales. It is relevant to this review as estuarine and coastal wetlands are captured in the method.

In the absence of quantitative information, the qualitative approach utilises expert opinion and existing descriptive, spatially-referenced, biophysical coastal and marine data sets and maps to delineate boundaries of inshore waters (MCRA Technical Group1998; Commonwealth of Australia 2006). As with IBRA, data available from the different jurisdictions varied. IMCRA v3.3 was an inshore regionalisation that identified 60 meso-scale regions (see below), which were defined using biological and physical information, including the distribution of demersal fishes, marine plants and invertebrates, sea floor geomorphology and sediments, and oceanographic data (Commonwealth of Australia 2006). The hierarchy of spatial units used in IMCRA is:

- >1000s of km macro-scale continental provinces,
- 100s-1000s of km meso-scale regions,
- 10s-100s of km micro-scale local units,
- <10 km pica-scale sites.

### 3.6.3 Multi-attribute River Typology: Turak and Koop (2008) - NSW

**Purpose:** Develop a multi-attribute river typology for NSW, with potential for application at larger extents.

**Use:** Condition assessment and conservation planning in river systems.

**Key features:** Quantitative; produces multiple typologies depending on available data sets using a bottom-up approach. River types were classified using data from 322 reference sites. For each group of attributes a different number of river types were identified. Classification tree analysis was used to define regions. Spatial unit for river reaches was roughly between 0.1 and 1 km lengths.

This method is a classification of river types, which includes a regionalisation. The spatial scale of the classification is river reaches of between 0.1 – 1 km lengths. The classification method uses regions as the basis for fine-tuning river typologies. Seven broad regions are used to represent biogeographic patterns across NSW, providing the initial spatial unit for selecting reference sites. The method is included as a holistic regionalisation, with three of the four typologies based on biota, the fourth on abiotic attributes. Different sets of attributes were used to create abiotic, fish, edge macroinvertebrate, and riffle macroinvertebrate datasets on which the typologies were based. In the process of identifying river types, Turak and Koop (2008) delineated geographic regions on the basis of discontinuities in the classification trees for each of the attribute datasets.

Turak and Koop (2008) suggested that the boundaries they used to aid in identifying river types provide an indication of what freshwater ecoregions within NSW may look like. They also argue that bottom-up approaches to river classifications based on the composition of biological data accurately reflect ecological heterogeneity within rivers. With regard to the present review and application at the national extent, a limitation of this approach is the need for adequate spatial coverage in biological datasets.

### 3.6.4 Abiotic and Biotic Estuary Regionalisation: Pease (1999) - NSW

**Purpose:** To determine spatial management units (regions) for estuarine commercial fisheries of NSW based on similarity of shared environmental and fisheries attributes.

**Use:** Identification of estuary regions for NSW fishery management.

**Key features:** Quantitative; initial regionalisation based on 8 primary physical attributes and subsequently correlated with fisheries attributes (catch data by taxa and catch effort). Three latitudinal regions were delineated. Use of commercial fisheries data is novel.

Data from 53 estuaries in NSW were used as the primary spatial units for the regionalisation. Eight physical/environmental attributes were chosen for the initial spatial classification based on being readily quantifiable and known to be linked with species distribution and diversity (Pease 1999) (see Table 14). Multivariate analysis showed that physical factors contributed most to the delineation of the three regions, particularly those related to latitude and estuary size. The commercial fisheries data used was annual mean catch per 81 taxa (fish and shellfish) and mean number of days fished per fisher for each of the 53 estuaries from a five year period.

**Table 14: Physical/environmental attributes used by Pease (1999) (for sources of data see original paper).**

| Attributes | Units |
|---|---|
| Latitude | Degrees and minutes |
| Geomorphological type | 1. Saline coastal lagoon<br>2. Barrier estuary<br>3. Drowned river valley estuary |
| Catchment area | Square kilometres |
| Water area | Square kilometres |
| Entrance depth | Meres |
| Entrance width | Metres |
| Average rainfall | Millimetres |
| Seagrass area | Square kilometres |

### 3.6.5 Waters of National Importance (WONI), New Zealand - Freshwater ecosystem biogeographic framework

**Purpose:** Aims to identify geographic units with similar physical disturbance regimes, whilst considering recolonisation pathways and barriers to the dispersal of freshwater biota.

**Use:** Identification of water bodies of national importance in New Zealand.

**Key features:** Qualitative; regionalisation is based on species distribution data of non-diadromous fish, genetic similarity of aquatic biota populations and physical disturbance regimes. Seven provinces and 29 units were defined.

New Zealand's Waters of National Importance (WONI) program forms part of their Sustainable Development Program of Action for Freshwater, which identifies WONI's for tourism, irrigation, energy generation, industrial uses, recreation, natural heritage and cultural heritage (Chadderton et al. 2004; Leathwick et al. 2007). Leathwick et al. (2007) produced a biogeographic framework for identifying high natural value freshwater ecosystems as part of this process.

Hydroclasses and a spatial framework that captures biogeographic patterns were considered essential elements in the identification of nationally-important freshwater ecosystems. The framework includes a nested approach, which allows the various hydroclasses to be assessed using criteria applied within each biogeographic unit (Leathwick et al. 2007). Environmental and biogeographical classifications were developed separately and then combined. Limited biological data meant that species data was only adequate for fish. Macroinvertebrate data were considered too limited as in the majority of cases only presence/absence and generic or family level data were available. The inclusion of analysis of genetic differences between groups of related fish species allowed inferences to be made regarding past geographic linkages between different water bodies. The focus of the analysis and classification was on delineating historic catastrophic disturbance and recolonisation, rather than on identifying differing physical environments or habitat (Leathwick et al. 2007). Regional boundaries were delineated using expert opinion; the data from the fish analysis were given the most weight, a reflection of the confidence and coverage in the fish data compared with the other biological data sets. Where possible, catchments were used as the basic geographical unit from which two levels of regions were developed: provinces and units (Leathwick et al. 2007).

### *3.6.6 Ecological Framework: Canada*

**Purpose:** Managing natural resources.

**Use:** Principle use was for reporting state of the environment and sustainability of ecosystems in Canada.

**Key features:** Qualitative; uses both an ecoregions approach in conjunction with a classification and is applied to terrestrial and marine systems in a single regionalisation. Terrestrial systems explicitly include inland aquatic ecosystems.

Marshall and Schut (1999) describe the ecological framework developed in Canada. Their approach evolved from the desire to provide a common hierarchical ecosystem framework and terminology, with the underlying principle of the initiative being to manage resources at the ecosystem level. A nationally agreed spatial context, within which ecosystems at various levels of generalization could be described and monitored, was developed in order to move from managing elements of ecosystems to management of whole ecosystems. Marshall et al. (1999) detail 26 attributes related to the cover area of each ecological unit (2 attributes); climate (14 attributes); physical landscape characteristics (8 attributes): land cover (1 attribute); and population (1 attribute). Definitions and the number of map units (in parenthesis) for the four levels of generalization are outlined below (modified from Marshall and Schut 1999):

- **Ecozone** (15): at the top of the hierarchy, it defines the ecological mosaic of Canada on a sub-continental scale. They represent an area of the earth's surface representative of large and very generalized ecological units characterized by interactive and adjusting abiotic and biotic factors. Canada is divided into 15 terrestrial ecozones.
- **Ecoprovince** (53): a subdivision of an ecozone characterized by major assemblages of structural or surface forms, faunal realms, and vegetation, hydrology, soil, and macro climate.
- **Ecoregion** (194): a subdivision of an ecoprovince characterized by distinctive regional ecological factors, including climate, physiography, vegetation, soil, water, and fauna.
- **Ecodistrict** (1021): a subdivision of an ecoregion characterized by distinctive assemblages of relief, landforms, geology, soil, vegetation, water bodies and fauna.

In British Columbia, Canada, two complementary regionalisations are used: the Ecoregion Classification and the Biogeoclimatic Ecosystem Classification (BEC). The Ecoregion Classification is a modification of the Ecological Framework and has been revised a number of times. It is used to stratify the terrestrial and marine ecosystems into discrete geographical units at five levels. *Ecodomains* and *ecodivisions* are the highest levels are very broad and place British Columbia in a global context. The lower levels of ecoprovinces, ecoregions and ecosections are progressively more detailed and narrow in scope and relate segments of

British Columbia to one another. They describe areas of similar climate, physiography, oceanography, hydrology, vegetation and wildlife potential (Demarchi 1996).

The BEC delineates biogeoclimatic units on the basis of vegetation, soils, and climate; ecosystems are classified further on the basis of climax or mature successional vegetation stages. BEC is predominantly used for conservation and in forestry applications (http://www.for.gov.bc.ca/hre/becweb/index.html accessed March 2008). It was developed to provide a systematic view of the small-scale ecological relationships in the Province, given its great ecological complexity. It is based on macroclimatic processes, and physiography, which is a fundamental difference between this and all other regional ecosystem classifications (Demarchi 1996). A unique aspect of BEC is the integration of terrestrial and marine environments, resulting in a single regional ecosystem classification (Demarchi 1996). BEC hierarchical levels have been defined as follows (Demarchi 1996):

1. **Ecodomain:** an area of broad climatic uniformity, defined at the global level;
2. **Ecodivision:** an area of broad climatic and physiographic uniformity, defined at the continental level;
3. **Ecoprovince:** an area with consistent climatic processes or oceanography, and relief, defined at the sub-continental level;
4. **Ecoregion:** an area with major physiographic and minor macroclimatic or oceanographic variation, defined at the regional level;
5. **Ecosection:** an area with minor physiographic and macroclimatic or oceanographic variation, defined at the sub-regional level.

The first two levels listed above are broad and place British Columbia in a global context. The last three levels are progressively more detailed and narrow in their scope and relate the Province to other parts of North America or the Pacific Ocean, or different segments of the Province to each other on the basis of similar climate, physiography, vegetation, and wildlife potential (Demrchi 1996). For the terrestrial environment, the last two levels can be further subdivided by biogeoclimatic criteria, allowing detailed interpretation of climate, topography, soil, and vegetation for habitat and wildlife management. For the marine environment, these two levels can subdivided by biophysical criteria to allow interpretation of climate, bathymetry, water chemistry, and currents for fisheries management (Demrchi 1996). Demarchi (1996) claims the main value of the system is that it will place any ecosystem in a local, regional, provincial, continental or global context.

## 3.7 Applicability of regionalisation for identifying HCVAE

### *3.7.1 Criteria for reviewing applicability of regionalisation approaches*

The purpose of this review is to summarise and critique bioregionalisation approaches for aquatic ecosystems with regard to their use at the national level for identifying HCVAE. Unfortunately the criteria for identifying HCVAE in Australia is yet to be finalised; critiques of bioregionalisation are, therefore, limited to being of a general nature. Criteria similar to those used to compare the classifications of aquatic ecosystems in section 2.4 (although there are some differences in interpretation of the criteria) have been used to consider the attributes of the various regionalisation methods discussed previously. The criteria used are:

1. **Purpose:** those developed for conservation planning are considered more relevant.
2. **Scientifically valid:** based on best available science.
3. **Inclusive:** can be applied to wetlands, rivers and estuarine systems.
4. **Comprehensive:** considered to have appropriate spatial extent/grain.
5. **Objective:** can be applied by different people with different values and experiences with the same end result.
6. **Ecologically meaningful:** results in regions with distinct ecological characteristics and based on aquatic ecosystems.
7. **Feasible:** can be applied with the current level of knowledge of aquatic ecosystems, or with minimal additional sampling effort (remote or on-ground) at the continental scale.
8. **Compatible:** can be integrated or overlain with current systems of regionalisation used at the national and State levels.

The biogeographic regionalisation methods described above have been assessed against each of these criteria in turn. In addition, a summary table showing the outputs of this review against all criteria is provided.

It should be noted that this is a desktop review based on the available literature and descriptions of biogeographic regionalisation systems. Actual testing of the various methods would be required in order to objectively apply most of the criteria listed above. The critique of each regionalisation system should only be considered as a general summary to guide the Aquatic Ecosystems Task Group in its deliberations, not as a definitive assessment.

Whilst there have been numerous calls in the scientific literature for comparisons of regionalisation approaches, few have been undertaken, with methods being dependent on the question of the comparison (e.g. matching boundaries, patterns of similarity of single attributes).

> *"Although often demanded, there are no generally accepted quantitative methods for comparing alternative ecoregion schemes. Because they simply represent alternative grouped arrangements, no set of ecoregions is ''true'' or ''wrong'' in a statistical sense. Just as the correct time cannot be established from a consensus among many watches, no single set of ecoregions can be judged objectively as superior. Nevertheless, qualitative expert-based ecoregions have become the accepted standard against which alternative ecoregion schemes are compared. A method for testing significant differences among alternative ecoregion maps would be valuable, yet statistics appropriate for the pairwise comparison of alternative categorical maps are lacking, particularly when those maps could contain different numbers of categories. Nor does having equal numbers of ecoregions necessarily improve a one-to-one comparison. Significance testing is exacerbated by the absence of appropriate null models for testing spatial pattern (but see Hargrove and others 2002). **More fundamentally, a search for consensus among inherently different regionalisations created for unique purposes might be neither appropriate nor justified.** A theoretical and technical framework for evaluating differences among ecoregions remains an open research question."* Hargrove and Hoffman (2004).

### 3.7.2 Purpose

From the discussion on the various regionalisations presented above, it can be seen that there are several purposes for which regionalisations can be established. Stating the purpose of a regionalisation is a critical step in selecting/developing a method to apply to the regionalisation. In this instance, the main goal is to develop a spatial framework which will advance the identification, classification and management of HCVAE. Whilst there can be subtle overlaps in purpose, for example water resource management could be interpreted to include consideration of conservation of valued ecosystems, the main concern is conservation planning. The methods which were specifically developed for conservation planning include:

- Global frameworks;
- Environmental Domains - Mackey et al. (2008);
- Landscape framework for rivers and streams - Stein (2007);
- TNC freshwater classification and ecoregions – Higgins et al. (2005);
- WWF freshwater ecoregions – Abell et al. (2002);
- IBRA – Thackway and Cresswell (1995);
- IMCRA;
- Multi-attribute River Typology – Turak and Koop (2008);
- WONI New Zealand – Leathwick et al. (2007).

### 3.7.3 Scientifically valid

Each of the main types of bioregionalisation identified could be used to establish a national spatial framework for identifying and managing HCVAE. In section 2.4.2 it was argued that classification approaches that use drivers or were holistic in nature were the most scientifically robust. A similar argument is made here; regionalisations, which are based on ecosystem drivers or are considered holistic are deemed to be more robust within the context of this review.

In the past biodiversity conservation efforts have focused on species endemism, rarity and threatened species, habitats, ecosystems or population dynamics (e.g. large breeding events). Factors that affect biological regionalisations include data limitations (spatial coverage and biases in location data), and assumptions that single species/taxon approaches are representative of broader biodiversity patterns (Pressey 2004; Darwall and Vié 2005; Proches 2008). To avoid biases in biological approaches it may be necessary to combine maps based on endemism and diversity with multi-taxon regionalisations (Proches 2008); the point being that biodiversity surrogates only provide part of the picture and so including more surrogates should give a clearer picture (Pressey 2004). Data limitations are believed to be a limiting factor for the applicability of biological approaches at the national level, although Tait et al. (2003) argued that Unmack's regionalisation could form the basis of an interim freshwater regionalisation at the national level.

Many consider conservation of ecological processes to be a more practical way of conserving biodiversity in the long run (Prosches 2008), as a process-based approach uses ecosystem drivers to describe regional patterns. An important principle in choosing an appropriate spatial framework is ensuring that the patterns represented by the regionalisation persist (Knight et al. 2006). Persistence in turn requires the maintenance of ecological processes, which can include ecological, evolutionary, geomorphological, and hydrological processes (Cowling et al. 1999; Knight et al. 2006).

The regionalisations considered to best meet the criterion of scientific validity are:

- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007);
- River environmental domain analysis – Jerie et al. (2003);
- Bailey – Omernik ecoregions;
- TNC freshwater classification and ecoregions – Higgins et al. (2005);
- WWF freshwater ecoregions – Abell et al. (2002); and

- IBRA – Thackway and Cresswell (1995);
- IMCRA;
- Multi-attribute River Typology – Turak and Koop (2008);
- Abiotic and biotic estuary regonalisation – Pease (1999)
- WONI New Zealand – Leathwick et al. (2007);
- Ecological framework Canada.

Proponents for and against qualitative and quantitative approaches are often equally vehement in their arguments. However, as there has been no directive received from DEWHA as to a preference of one approach over the other, no judgment is made regarding the suitability of any of the approaches or methods available. The choice of approach is likely to be governed by many factors and the integration of elements of qualitative approaches, in particular expert input, and quantitative techniques will most likely become more common. IMCRA is one such system that explicitly acknowledges this type of approach.

Qualitative methods include:

- Global frameworks;
- Macroinvertebrate regions Victoria;
- Freshwater Bioregionalisation of QLD riverine ecosystems;
- Environmental Domains - Mackey et al. (2008);
- Bailey – Omernik ecoregions;
- TNC freshwater classification and ecoregions – Higgins et al. (2005);
- WWF freshwater ecoregions – Abell et al. (2002);
- Primary productivity regimes – Mackey et al. (2008).
- IBRA – Thackway and Cresswell (1995);
- IMCRA;
- WONI New Zealand – Leathwick et al. (2007);
- Ecological framework Canada.

Quantitative methods include:

- Freshwater fish – Unmack (2001);
- Freshwater fish – Growns and West (in review);
- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007);
- River environmental domain analysis – Jerie et al. (2003);
- Multi-attribute River Typology – Turak and Koop (2008);
- Abiotic and biotic estuary regionalisation – Pease (1999).

### *3.7.4 Inclusive*

Most regionalisations are not inclusive of all aquatic ecosystem types, although most of those reviewed here do give consideration to particular aquatic ecosystems. IBRA and IMCRA combined could be considered inclusive as combined they capture the majority of aquatic ecosystems; however they do not as stand alone frameworks. To address this will require either a regionalisation system purpose-built to capture all aquatic ecosystems, or for a system where the outputs of terrestrial, riverine, and coastal regionalisations are overlain and expert opinion used to delineate ecologically meaningful boundaries.

Only one regionalisation reviewed, the Ecological Framework of Canada, includes both terrestrial and marine ecosystems in the one framework. Whilst rivers and river dependent ecosystems are considered in many approaches, non-riverine aquatic ecosystems such as karsts, groundwater dependent ecosystems, saltmarshes and mangroves are not well represented. Relatively few studies have directly investigated the efficacy of regionalisations to group lakes (non-riverine wetlands) (Cheruvelil et al. 2008), with Jenerette et al. (2002 cited Cheruvelil et al. 2008) finding ecoregions only 18% effective at classifying lakes (based on 15 water chemistry and quality variables). Adequate representation of intertidal habitats requires that design and planning for marine and terrestrial protected areas be done together, with intertidal habitats being explicitly targeted (Banks et al. 2005).

The only regionalisation which met this criterion was the Canadian system.

### 3.7.5 Comprehensive

With regard to regionalisations, the criterion of comprehensiveness is taken to relate to extent (the overall area of a study) and grain (the lowest resolution or minimal mapping unit), as these features define the upper and lower limits of resolution used (McMahon et al. 2004).

In order to better include the requirements for protecting comprehensive, adequate and representative (CAR) samples of aquatic ecosystems, the national framework must flexible in order to be applied at multiple scales (e.g. regional, State and national extents). Regionalisations developed at the State level (e.g. Victorian bioregions, Turak and Koop 2008, QLD bioregions) would require considerable refinement before being applied at the national level, despite claims of some methods being applicable at any scale; if the extent is changed then the regions are also likely to change. The detection of pattern is a function of the scale and variability of information relating to the entity being classified. There is no single correct scale for regionalisation (McMahon et al. 2004) and so all methods could be considered comprehensive within the bounds of purpose for which each was designed.

The framework to identify HCVAE will be a national framework and therefore this is considered the relevant extent to meet this criterion. Regionalisations that can be applied at the relevant extent (continental) include:

- Freshwater fish – Unmack (2001);
- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007);
- Primary productivity regimes – Mackey et al. (2008)
- IBRA – Thackway and Cresswell (1995);
- IMCRA.

### 3.7.6 Objective

The comments made in section 2.4.5 and 3.7.3 are relevant here as well; with the principle issue considered here being identification of regional boundaries. What constitutes a boundary and how they are delineated is a major area of discussion/debate in the scientific literature on ecological regionalisations (McMahon et al. 2004; Mackey et al. 2008) and relates to the choices between qualitative and quantitative methods.

The critical point relevant to this criterion is that boundaries are able to be identified consistently. This requires the decision rules to be clearly documented. Without testing the methods describe to see how consistently regional boundaries can be delineated all methods are considered to meet this objective.

### 3.7.7 Ecologically meaningful

In general, most definitions of ecological regions refer to ecosystems as the basic unit being grouped or classified (MacMahon et al. 2004). However, agreeing on what constitutes an ecologically meaningful regionalisation approach is open for debate as there is a lack of agreement on the definition of *ecosystems* (Omernik 2004). The difficulty lies with the inherent complexity of ecosystems: they are not static entities, they have multiple attributes/components and processes that vary both temporally and spatially (Omernik 2004). It follows then that it is very difficult to agree on what constitutes an ecologically meaningful region. Boundaries are difficult to agree on when all ecosystems at all spatial and temporal scales are open to some degree (MacMahon et al. 2004).

Bailey (2005) argues that climate is the major controlling factor in defining ecoregions and ecosystem boundaries, and that it should follow that ecoregions should reflect significant differences in climate. He provides a list of 20 principles that act as both descriptions of ecoregions, points for interpreting data requirements, and test points for analysing sites in the field. It is not possible to apply these principles here, but they are reproduced so as to provide the basis of further discussion/testing:

1. The series of ecoregions should express the changing nature of the climate over large areas.
2. Boundaries of ecoregions coincide with certain climatic parameters.
3. Fine-scale climatic variations can be used to delineate smaller ecological regions.
4. Boundaries should capture the effect of mountains on climate
5. A uniform pattern of mountain zonation is repeated over a climatic zone, which is the basic element in regionalizing mountainous territories.
6. Ecoregional boundaries should delineate groups of upland sites with similar characteristics.
7. The mosaic of ecosystems found in major transitional zones (ecotones) should be delineated as separate ecoregions.
8. Context is often important as content in mapping ecological regions, depending on scale.
9. Because subsystems can be understood only with the context of the whole, a classification of ecoregions begins with the largest unit and successively subdivides them.
10. The factors used to recognise ecoregions should be relatively stable.
11. Boundaries should circumscribe large, contiguous areas.
12. Potential vegetation, in contrast to actual or real, vegetation is useful in capturing ecological regions.
13. An understanding of the relationships between successions on identical landform positions in different comates is useful or establishing meaningful ecological regions.
14. Geological factors might modify zonal boundaries.
15. Establishing a specific hierarchy of ecoregional boundaries should be based on understanding the formative processes that operate to differentiate ecoregions at various scales.
16. Criteria for setting ecoregion boundaries should be explicit in how regions are identified on the basis of comparable likenesses and differences.
17. The limits of geographic ranges of species and races of plants and animals are not fully satisfactory criteria for determining boundaries of ecoregions.
18. Ecoregions should have greater ecological relevance than large physiographic land units.
19. Ecoregion boundaries should have greater ecological relevance than watersheds (or basins or hydrological units).
20. The boundaries of ecoregions emerge from the study of spatial coincidences, patterning, and relationships, of climate, vegetation, soil and landform.

The importance of climate has been acknowledged in several Australian studies. For example Hutchinson et al. (2005) integrated a global agro-climatic classification with the IBRA 5.1 boundaries, with the combined classification able to reflect major potential differences in landscape function and response to management. The approach of Hutchinson et al. (2005) and Hobbs and McIntyre (2005) were recently used to investigate the possible future impacts of climate change on the NRS (Dunlop and Brown 2008). Mackay et al. (2008) also produced a climate classification with their environmental domain analysis regionalisation.

Any classification or regionalisation involves assigning artificial boundaries to ecosystems (see section 2.4.6). Many of the regionalisations reviewed here do not explicitly define ecosystem or the context in which it is used. It could be argued that they are all ecologically meaningful within the context of their purpose; however those which refer specifically to aquatic ecosystems (e.g. rivers, riverine dependent ecosystems, etc) are taken to be the most relevant for the purpose of this review. Whilst some IBRA regions are characterised by aquatic ecosystems, for example Darling Riverine Plain, Riverina, Gulf Plains, and Swan Coastal Plain, IBRA is predominantly a terrestrial system.

Regionalisations which meet this criterion include:

- Freshwater fish – Unmack (2001);
- Freshwater fish – Growns and West (in review);

- Macroinvertebrate regions Victoria;
- Freshwater Bioregionalisation of QLD riverine ecosystems;
- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007);
- River environmental domain analysis – Jerie et al. (2003);
- Bailey – Omernik ecoregions;
- TNC freshwater classification and ecoregions – Higgins et al. (2005);
- WWF freshwater ecoregions – Abell et al. (2002);
- IMCRA;
- Multi-attribute River Typology – Turak and Koop (2008);
- Abiotic and biotic estuary regionalisation – Pease (1999)
- WONI New Zealand – Leathwick et al. (2007);
- Ecological framework Canada.

### *3.7.8 Feasible*

This criterion relates to data limitations and the ease with which the regionalisation could be applied at the continental extent. Bottom-up (section 2.4.7) or data-driven approaches that rely on species distribution data will have limitations associated with spatial coverage and biases in data sets (Pressey 2004), particularly in areas with little or no inventory data. As for the 'ecologically meaningful' criterion, it is important to consider the purpose and extent for which the regionalisations were developed when considering feasibility. Most regionalisations reviewed are not at the appropriate extent for identifying HCVAE at the national extent and would therefore require additional acquisition of data. Many make statements about applicability at larger extents (e.g. Turak and Koop 2008), but are yet to be tested. It should be noted that such data limitations do not in any way imply bottom up approaches are less valid, purely that there is often insufficient data available (e.g. Marshall et al. 2006).

Within the context of the current review, this criterion was considered to be best met by regionalisations which were top-down and at the continental extent:

- Freshwater fish – Unmack (2001);
- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007);
- IBRA – Thackway and Cresswell (1995);
- IMCRA.

### *3.7.9 Compatible*

As the national framework for identifying HCVAE will be required to apply at multiple scales and be relevant to the CAR principles of the NRS, it should be compatible with the existing frameworks used by the NRS (IBRA and IMCRA). A complete analysis of the compatibility of each of the regionalisations used in Australia with the IBRA and IMCRA regionalisations is beyond the scope of this review. It is likely, however, that the development of a purpose-built framework will be needed. Frameworks that have a continental extent and incorporate surface hydrological units as part of their spatial framework are considered likely to be the most compatible with the existing IBRA and IMCRA frameworks, and the overall objectives of the proposed framework for identifying HCVAE. Such frameworks include:

- Environmental Domains - Mackey et al. (2008);
- Drainage basins: Australian Water Resource Council;
- Landscape framework for rivers and streams - Stein (2007).

### 3.7.10 Summary

A summary of each of the regionalisation approaches considered in this review against the eight criteria is provided in Table 15.

**Table 15: Summary of criteria against regionalisation approaches. Note that this has been a subjective assessment.**

| Regionalisation system | Criteria for review | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
| **Biological approaches** | | | | | | | | |
| Global frameworks (Damann-Urdvary, Olson et al. 2001) | X | X | | | X | | | |
| Freshwater fish – Unmack (2001) | | X | | X | X | X | X | |
| Freshwater fish – Growns and West (in review) | | | | | X | X | | |
| Macroinvertebrate regions Victoria | | | | | X | X | | |
| Freshwater Bioregionalisation of QLD | | | | | X | X | | |
| **Ecosystem driver approaches** | | | | | | | | |
| Environmental Domains - Mackey et al. (2008) | X | X | | X | X | X | X | X |
| Drainage basins: Australian Water Resource Council | | X | | X | X | X | X | X |
| Landscape framework for rivers - Stein (2007) | X | X | | X | X | X | X | X |
| River environmental domain analysis – Jerie et al. (2003) | | X | | | X | X | | |
| Bailey – Omernik ecoregions | | X | | | X | X | | |
| TNC freshwater classification and ecoregions – Higgins et al. (2005) | X | X | | | X | X | | |
| WWF freshwater ecoregions – Abell et al. (2002) | X | X | | | X | X | | |
| **Ecosystem response approaches** | | | | | | | | |
| Primary productivity regimes – Mackey et al. (2008) | | | | X | X | | | |
| **Holistic approaches** | | | | | | | | |
| IBRA | X | X | | X | X | | X | X |
| IMCRA | X | X | | X | X | X | X | X |
| Multi-attribute River Typology – Turak and Koop (2008) | X | X | | | X | X | | |
| Abiotic and biotic estuary regionalisation – Pease (1999) | | X | | | X | | | |
| WONI New Zealand – Leathwick et al. (2007) | X | X | | | X | X | | |
| Ecological framework Canada | | X | X | | X | X | | |

Key to criteria:

1. **Purpose:** those developed for conservation planning are considered more relevant.
2. **Scientifically valid:** based on best available science.
3. **Inclusive:** can be applied to wetlands, rivers and estuarine systems.
4. **Comprehensive:** considered to have appropriate spatial extent/grain.
5. **Objective:** can be applied by different people with different values and experiences with the same end result.
6. **Ecologically meaningful:** results in regions with distinct ecological characteristics and based on aquatic ecosystems.
7. **Feasible:** can be applied with the current level of knowledge of aquatic ecosystems, or with minimal additional sampling effort (remote or on-ground) at the continental scale.
8. **Compatible:** can be integrated or overlain with current systems of regionalisation used at the national and State levels.

# 4 CONCLUSIONS

The project brief and discussions with DEWHA indicated that the outputs of this paper should not include any recommendations for specific classification or bioregionalisation schemes for aquatic ecosystems in Australia. The decision on an appropriate system(s) for identifying HCVAE is the responsibility of the Aquatic Ecosystems Task Group and the role of this paper is to provide the group with one source of information on which to base their decision.

However, the author's feel that there are a number of general messages that can be concluded from this review, and in the interest of completeness these are presented here.

**Finalisation of criteria for identifying HCVAE**
As evidenced from this review there are a number of different approaches for classification and bioregionalisation systems all with their strengths and weaknesses. Central to the choice of an appropriate classification and bioregionalisation system will be the characteristics that they need to represent with respect to HCVAE. Therefore it is imperative that the criteria for identifying HCVAE be finalised prior to the selection of an appropriate classification and bioregionalisation system(s).

**Testing of classification and bioregionalisation approaches**
The critique of classification and bioregionalisation methods provided above is based on expert opinion only. There was no opportunity to empirically test the systems with respect to feasibility of implementation across Australia or compatibility with systems currently in use. In the absence of criteria for identifying HCVAE the critique was limited to the general principles of comprehensiveness, adequacy and representativeness. Once criteria for identifying HCVAE have been finalised, objective testing of different methods would provide additional information in the development and implementation of a classification and bioregionalisation system(s).

**Purpose built**
Regardless of the Aquatic Ecosystem Task Group's ultimate decision for an approach to classification and bioregionalisation (e.g. top-down, bottom-up, qualitative, quantitative, etc.) it is unlikely that an existing system will meet all of the needs of a robust classification and bioregionalisation system for implementation across Australia. Systems used internationally and in different jurisdictions in Australia, while having their basis in a specific type of method, generally have required modification and adaption both to suit the purpose of the system and the types of aquatic ecosystems that are required to be covered.

**Build on existing information**
While it is true that a classification and bioregionalisation system for identifying HCVAE is likely to require some specific development, there is no need to start at the beginning. Considerable work has been undertaken in this field and guidance can be taken from the literature with respect to development of classification systems and the suitability of classification and bioregionalisation systems for identifying HCVAE.

**Compatibility**
There are a large number of classification and bioregionalisation systems already in use across Australia. A large amount of resources have been invested in the application of these various systems for inventory and conservation planning. The overwhelming view from stakeholder consultation was that it is unlikely that states and agencies will abandon the systems that are currently in use. As a consequence the system adopted at the national level will need to be compatible with those that are currently in place.

**Knowledge gaps**
A key observation was that whilst estuarine and non riverine wetlands are adequately covered in the various classification schemes reviewed, very few Australian examples of regionalisations which explicitly capture these aquatic ecosystem types were identified. Groundwater dependent ecosystems and karst ecosystems are also poorly represented.

# 5 REFERENCES

Abell, R., Thieme, M., Dinerstein, E., and Olson, D. (2002). A Sourcebook for Conducting Biological Assessments and Developing Biodiversity Visions for Ecoregion Conservation. Volume II: Freshwater Ecoregions. World Wildlife Fund, Washington, DC, USA.

Abell, R., Allan, J.D. and Lehner, B. (2007). Unlocking the potential of protected areas for freshwaters. *Biological Conservation* 134: 48–63.

Agences de L'Eau (1998). *SEQ-physique. Système d'evaluation de la qualité du milieu physique des cours d'eau*

Allan J.D. and Flecker A.S. (1993). Biodiversity conservation in running waters: identifying the major factors that threaten destruction of riverine species and ecosystems. *BioScience* 43: 32–43.

Australian Surveying and Land Information Group. (1990). Vegetation. Atlas of Australian Resources. AUSMAP, Department of Administrative Services, Canberra.

Bailey, R.G. (1983 ). Delineation of ecosystem regions. Environmental Management 7: 365-373.

Bailey, R.G., S.C. Zoltai, and E.B. Wiken. (1985). Ecological regionalization in Canada and the United States. *Geoforum* 116: 265-275.

Bailey, R.G. (2005). Identifying ecoregion boundaries. *Environmental Management* 34: S1-S26.

Ball, D., Blake, S. and Plummer, A. (2006). *Review of Marine Habitat Classification Systems*. Parks Victoria Technical Series No. 26. Parks Victoria, Melbourne

Banks, S.A., Skilleter, G.A. and Possingham, H.P. (2005). Intertidal habitat conservation: identifying conservation targets in the absence of detailed biological information. *Aquatic Conservation Marine and Freshwater Ecosystems* 15: 271-288.

Brierley, G.J. and Fryirs, K.A. (2005). *Geomorphology and River Management: Applications of the River Styles Framework.* Blackwell Publications, Oxford, UK.

Brierley, G., Fryirs, K., Outhet, D. and Massey, C. (2002). Application of the River Styles framework as a basis for river management in New South Wales, Australia. *Applied Geography* 22: 91-122.

Brinson, M.M. (1993). *A Hydrogeomorphic Classification for Wetlands*, Technical Report WRP–DE–4, U.S. Army Corps of Engineers Engineer Waterways Experiment Station, Vicksburg, MS.

Brooks, A., Brunner, P. and Lin, P. (2005). Classifying the diversity of Gulf rivers. RipRap Edition 28, Land and Water Australia
http://www.rivers.gov.au/Publications_and_Products/Rip_Rap/28._Tropical_Rivers/index.aspx

Bryan, B.A. (2006). Synergistic techniques for better understanding and classifying the environmental structure of landscapes. *Environmental Management* 37:126-140.

Bryce, S.A. and Clarke, S.E. (1996). Landscape level ecological regions: linking state level ecoregion frameworks with stream habitat classifications. *Environmental Management* 20: 297-311.

Bunce, R.G.H., Carey, P.D., Elena-Rossello, R., Orr, J., Watkins, J. and Fuller, R. (2002).A comparison of different biogeographical classifications of Europe, Great Britain and Spain. *Journal of Environmental Management* 65: 121-134.

Burgess, N.D., D'Amico Hales, J., Ricketts, T.H. and Dinerstein, E. (2006). Factoring species, non-species values and threats into biodiversity prioritisation across the ecoregions of Africa and its islands. *Biological Conservation* 127: 383-401.

Butcher, R. J. (2003). Assessing biodiversity in temporary and permanent freshwater wetlands in Victoria, Australia. PhD thesis, unpublished, Monash University.

Calvert, S., Erskine, W. and Junor, B. (2001) *Development of a National River Classification System - Final Report.* Consultancy report prepared by PPK Environment and Infrastructure. Murray-Darling Basin Commission, Environment Australia and the Land and Water Resources, Research and Development Corporation, Sydney, Australia

Chadderton, W.L., Brown, D.J., and Stephens, R.T. (2004). Identifying freshwater ecosystems of national importance for biodiversity: Criteria, methods, and candidate list of nationally important rivers. Department of Conservation, Wellington, New Zealand.

Cheruvelil, K.S., Soranno, P.A., Bremigan, M.T., Wagner, T. and Martin, S.L. (2008). Grouping lakes for water quality assessment and monitoring: the roles of regionalization and spatial scale. *Environmental Management* 41:425-440.

Commonwealth of Australia (2006). A Guide to the Integrated Marine and Coastal Regionalisation of Australia Version 4.0. Department of the Environment and Heritage, Canberra, Australia.

Conrick, D.L., Edgar, B. and Innes, A. (in prep). *Development of National Indicators for Wetland Ecosystem Extent, Distribution and Condition.* Final report. NLWRA, Canberra.

Corrick, A.H., and Norman, F.I. (1980). Wetlands of Victoria I. Wetlands and waterbirds of the Snowy River and Gippsland Lakes catchment. *Proceedings of the Royal Society of Victoria* 91:1–15.

Cowardin, L.M., Carter, V., Golet, F.C. and LaRoe, E.T. (1979). *Classification of Wetlands and Deepwater Habitats of the United States*. US Department of the Interior, Fish 7 Wildlife Service, Office of Biological Services, Washington DC 20240

Cowardin, L.M. and Golet, F.C. (1995). U.S. Fish and Wildlife Service 1979 wetland classification - a review. *Vegetatio* 118: 139-152.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G. and Ellis, A.G. (1999). From representation to persistence: requirements for a sustainable reserve system in the species rich mediterranean-climate deserts of southern Africa. *Diversity and Distributions* 5:51-71.

Crowley, J.M. (1967). Biogeography. Canadian Geographer 11:312-326.

Darwall, W.R.T. and Vié, JC. (2005) Identifying important sites for conservation of freshwater biodiversity: extending the species-based approach. *Journal of Fisheries Management and Ecology* 12:287-293.

Demarchi, D.A (1996). An Introduction to the Ecoregions of British Columbia. Wildlife Branch Ministry of Environment, Lands and Parks Victoria, British Columbia. http://www.env.gov.bc.ca/ecology/ecoregions/index.html

Department of Natural Resources, Mines and Water (2006). Freshwater bioregionalisation of Queensland's riverine ecosystems, Summary. Queensland Government.

Digby, M. J., Saenger, P., Whelan, M. B., McConchie, D., Eyre, B., Holmes, N. and Bucher, D. (1999). *A Physical Classification of Australian Estuaries*. National River Health Program – Project E1. Urban Sub-Program Report No.9. Occasional Paper 16/99, Land and Water Resources Research and Development Corporation

Dini, J., Cowan, G. and Goodman, P. (1998). Proposed Wetland Classification System for South Africa, South African National Wetland Inventory, http://www.ngo.grida.no/soesa/nsoer/resource/wetland/inventory_classif.htm

DPIW (Department of Primary Industry Water and Environment) (2007). Auditing Tasmania's Freshwater Ecosystem Values. Summary of assessment framework A strategic framework for statewide management and conservation of Tasmania's freshwater ecosystem values. Water Assessment Branch, Water Resources Division, DPIW, Tasmania.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Zl., Knowler, D.J., Lévêque, C., naiman, R.J., Prieur_Richard, AH., Soto, D., Stiassny, M.L.J. and Sullivan, C.A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* 81: 163–182.

Duguid, A., Barnetson, J., Clifford, B., Pavey, C., Risler, J. and McNellie, M. (2002). *Wetlands in the arid Northern Territory*. Report to Environment Australia on the inventory and significance of wetlands in the arid NT. Parks and Wildlife Commission of the Northern Territory. Alice Springs

Dunlop, M., and Brown, P.R. (2008). Implications of climate change for Australia's National Reserve System: A preliminary assessment. Report to the Department of Climate Change, February 2008. Department of Climate Change, Canberra, Australia.

Eekhout, S., King, J.M. and. Wackernagel, A. (1997). Classification of South African Rivers. Department of Environmental Affairs and Tourism, Pretoria

Environmental Protection Agency (2005). *Wetland mapping and Classification Methodology: Overall Framework – A method to provide Baseline mapping and classification for wetlands in Queensland*. Version 1.2. Queensland Government, Brisbane

Farinha, J.C., Costa, L., Zalidis, G., Mantzavelas, A., Fitoka, E., Hecker, N. and Vives, P.T. (1996). Mediterranean Wetland Inventory: Habitat Description System. MedWet / Instituto da Conservacao da Natureza (ICN) / Wetlands International / Greek Biotope / Wetland Centre (EKBY) Publication Volume III.

Finlayson, M. (2006). Freshwater protected areas: can we expand our options to include private wetlands? *Ecological Management and Restoration* 7: 77-78.

Fitzsimons, J.A. and Robertson, H.A. (2005). Freshwater reserves in Australia: directions and challenges for the development of a comprehensive, adequate and representative system of protected areas. *Hydrobiologia* 522: 87–97.

Fieseler, C. and Wolter, C. (2006). A fish based typology of small temperate rivers in the northeastern lowlands of Germany*. Limnologica* 36: 2–16

Finlayson, C.M. and van der Valk, A.G. (1995). Wetland classification and inventory: A summary. *Vegetatio* 118: 185-192

Frissell, C. A., Poff, N. L. and Jensen, M. E. (2001). Assessment of biotic patterns in freshwater ecosystems. In: Jensen, M.E. and Bourgeron, P. S. (eds.) *A Guidebook for Integrated Ecological Assessment*, Springer-Verlag, New York

Green, D. (1997). *Wetland Management Technical Manual: Wetland Classification*. Environmental Studies Unit. January 1997.

Growns I (in review) Differences in bioregional classifications among four aquatic biotic groups: implications for conservation reserve design and monitoring programs.

Growns I and West G (in review) An objective approach to the classification of aquatic bioregions through the use of distributional modelling of freshwater fish.

Hargrove, W.W. and Hoffman, F.M. (2004). Potential of multivariate quantitative methods for delineation and visualization of ecoregions. *Environmental Management* 34: S39-S60.

Hauer, F.R. and Smith, R.D. (1998). The hydrogeomorphic approach to functional assessment of riparian wetlands: evaluating impacts and mitigation on river floodplains in the USA. *Freshwater Biology*, 40: 517-530.

Hawkins, C.P. and Vinson, M.R., (2000). Weak correspondence between landscape classifications and stream invertebrate assemblages: implications for bioassessment. *Journal of the North American Benthological Society* 19: 501–517.

Higgins, J. V., Bryer, M. T., Khoury, M. L. and FitzHugh, T. W. (2005). A freshwater classification approach for biodiversity conservation planning. *Conservation Biology* 19: 432-445.

Hobbs, R.J. and McIntyre, S. (2005) Setting limits to generalization: a framework for considering landscape characteristics across Australia. *Global Ecology and Biogeography*, 14, 1–5.

Hurley, M. and Jensen, M.E. (2001). Geomorphic patterns, processes and perspectives in aquatic assessment. In: Jensen, M.E. and Bourgeron, P. S. (eds.) *A Guidebook for Integrated Ecological Assessment*, Springer-Verlag, New York

Hutchinson, M, F., McIntyre, S., Hobbs, R.J., Stein, J.L., Garnett, S. and Kinloch, J. (2005). Integrating a global agro-climatic classification with bioregional boundaries in Australia. *Global Ecology and Biogeography* 14: 197-212.

Interim Marine and Coastal Regionalisation for Australia (IMCRA) Technical Group. (1998). Interim Marine and Coastal Regionalisation for Australia: an ecosystem- based classification for marine and coastal environments. Version 3.3. Environment Australia, Commonwealth Department of the Environment. Canberra.

Jensen, M.E., Goodman, I.A., Frissell, C.A., Brewer, C.K. and Bourgeron, P.S. (2001). Ecological classification and mapping of aquatic systems. In: Jensen, M.E. and Bourgeron, P. S. (eds.) *A Guidebook for Integrated Ecological Assessment*, Springer-Verlag, New York

Jepson, P. and Whittaker, R.J. (2002). Ecoregions in context: a critique with special reference to Indonesia. *Conservation Biology* 16: 42-57.

Jerie, K and Houshold, I. (2003). Report on the project to identify river landscapes for the Conservation of Freshwater Ecosystem Values project. November 2003. Supplied by DPIW Tasmania.

Jerie, K., Houshold, I. and Peters, D. (2003). Tasmania's river geomorphology: stream character and regional analysis. Nature Conservation Report 03/5. Nature Conservation Branch, Department of Primary Industries, Water and Environment, Hobart. (Volume 1 and Volume 2: Appendices)

Johnson, P. and Gerbeaux, P. (2004). *Wetland Types in New Zealand*. Department of Conservation, Te Papa Atawhai, PO Box 10-420, Wellington.

Kent, D. (2000). Identification, classification and US Regulation. In: Kent, D. (ed.) *Applied Wetlands Science and Technology*, CRC Press, Florida USA.

Kingsford, R.T., Dunn, H., Love, D., Nevill, J., Stein J. and Tait, J. (2005). *Protecting Australia's rivers, wetlands and estuaries of high conservation value: a blueprint*. Report to Land and Water Australia, Canberra. http://www.environment.gov.au/water/publications/environmental/protectingrivers.html.

Kingsford, R.T. and Neville, J. (2005). Scientists urge expansion of freshwater protected areas. *Ecological Management & Restoration* 6: 161–162.

Knight, A.T., Driver, A., Cowling, R.M., Maze, K., Desmet, P.G., Lombard, A.T., Rouget, M., Botha, Boshoff, A.F., Castley, J.G., Goodman, P.S., Mackinnon, K., Pierce, S.M., Sims-Catley, R., Stewart, W.I., and von Hase, A. (2006). Designing systematic conservation assessments that promote effective implementation: best practice from South Africa. *Conservation Biology* 20: 739-750.

Larmour, G. (2001). Wetland classification system. Criteria for inclusion and data presentation. In: *A Directory of Important Wetlands in Australia*. 3rd edition. Environment of Australia, Canberra.

Leathwick, J.R., Collier, K. and Chadderton, L. (2007). Identifying freshwater ecosystems with nationally important natural heritage values: development of a biogeographic framework. Science for Conservation 274. Published by Science and Technical Publishing, Department of Conservation, Wellington, New Zealand.

Leathwick, J.R., Overton, J. McC., and McLeod, M. (2003). An environmental domain classification of New Zealand and its use as a tool for biodiversity management. *Conservation Biology* 17: 1612-1623.

Linke, S. (2006). River conservation planning: accounting for condition, vulnerability and connected systems. PhD Thesis, University of Canberra.

Linke, S., Pressey, R.L., Bailey, R.C. & Norris R.H. (2007). Management options for river conservation planning: condition and conservation revisited. *Freshwater Biology* 52: 918–938.

Loveland, T. R. and Merchant, J.M. (2004). Ecoregions and ecoregionalization: geographical and ecological perspectives. *Environmental Management* 34: S1–S13.

McMahon, G., Gregonis, S.M., Waltman, S.W., Omernik, J.M., Thorson, T.D., Freeouf, J.A., Rorick, A.H. and Keys, J.E. (2001). Developing a spatial framework of common ecological regions for the conterminous United States. *Environmental Management* 28:293-316.

McMahon, G., Wiken, E.B. and Gauthier, D.A. (2004). Toward a scientifically rigorous basis for developing mapped ecological regions. *Environmental Management* 34: S111-S124.

Mackey, B.G., Berry, S.L. and Brown, T. (2008). Reconciling approaches to biogeographical regionalisation: a systematic and generic framework examined with a case study of the Australian continent. *Journal of Biogeography* 35: 213-229.

Marchant, R., Ryan, D. and Metzeling, L. (2006). Regional and local species diversity patterns for lotic invertebrates across multiple drainage basins in Victoria. *Marine and Freshwater Research* 57: 675–684.

Marchant, R., Wells, F. and Newall, P. (2000). Assessment of an ecoregion approach for classifying assemblages from streams in Victoria, Australia. *Journal of the North American Benthological Society* 19: 497–500.

Margules, C.R. and Pressey, R.L. (2000). Systematic conservation planning. Nature, 405, 243–253.

Marshall, I.B and Schut, P.H (1999). A national ecological framework for Canada – Overview. A cooperative product by Ecosystems Science Directorate, Environment Canada and Research Branch, Agriculture and Agri-Food Canada.

Marshall, I.B., Schut, P.and M. Ballard (compilers). (1999). *A National Ecological Framework for Canada: Attribute Data.* Environmental Quality Branch, Ecosystems Science Directorate, Environment Canada and Research Branch, Agriculture and Agri-Food Canada, Ottawa/Hull. http://sis.agr.gc.ca/cansis/nsdb/ecostrat/data_files.html#summary_table

Marshall, J., Prior, A., Steward, A. and McGregor, G. (2006). Freshwater bioregionalisation of Queensland's riverine ecosystems. Development of interim freshwater biogeographic provinces. Aquatic Ecosystems Technical Report No. 56, Queensland Department of Natural Resources, Mines and Water.

Metzeling, L. Tiller, D., Newall, P., Wells F. and Reed J. 2006). Biological objectives for the protection of rivers and streams in Victoria, Australia. *Hydrobiologia* 572: 287-299.

Metzeling, L., Wells, F. and Newall, P. (2002). The development and testing of biological objectives for the protection of rivers and streams in Victoria, Australia. *Verhandlungen Internationale Vereinigung Limnologie* 28: 106–111.

Moilanen, A., Leathwick, J. and Elith, J. (2008). A method for spatial freshwater conservation. *Freshwater Biology* 53: 577-592.

Morgan, G. and Terrey, J. (1992). Nature Conservation in Western New South Wales. NSW National Parks Association, Sydney.

Naiman, R.J., Lonzarich, D.G., Beechie, T.J. and Ralph, S.C. (1992). General principles of classification and the assessment of conservation potential in rivers. In: Boon, P. J., Calow, P.and Petts, G. E. (eds.) *River Conservation and Management.* John Wiley, Chichester, pp. 93–124.

National Resource Information Centre and CSIRO Division of Soils (1991). Digital Atlas of Australian Soils. Canberra.

Natural Resource Management Ministerial Council (2005). Directions for the National Reserve System – A Partnership Approach. Commonwealth of Australia. http://www.deh.gov.au/parks/nrs/index.html

Nel, J.L., Roux, D.J., Marree, G., Kleynhans, C.J., Moolan, J., Reyers, B., Rouget, M. and Cowling, R.M. (2007). Rivers in peril inside and outside protected areas: a systematic approach to conservation assessment of river ecosystems. *Diversity and Distribution* 13: 341-352.

Nevill, J. (2005). Urgent need for a systematic expansion of freshwater protected areas in Australia: A scientists consensus statement. OnlyOnePlanet Australia, Hampton, Melbourne. www.onlyoneplanet.com.au

Nevill J (2006). 'Freshwater protected areas in Australia', paper prepared for the 2006 Australian State of the Environment Committee, Department of the Environment and Heritage, Canberra, http://www.environment.gov.au/soe/2006/publications/emerging/freshwater/pubs/freshwater.pdf

Nevill, J. and Phillips, N. (eds) (2004). *The Australian Freshwater Protected Area Resourcebook: the policy background, role and importance of protected areas for Australian inland aquatic ecosystems.* OnlyOnePlanet Australia; Hampton Melbourne.

Olson, D., Chernoff, B., Burgess, G., Davidson, I. & Canevari, E. (1997). Freshwater biodiversity of Latin America and the Caribbean: a conservation assessment. Washington D.C.: World Wildlife Fund.

Olson, D.M., Dierstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P. and Kassem, K.R. (2001). Terrestrial ecoregions of the world: a new map of life on earth. *BioScience* 51: 933-938.

Omernik, J.M (1987). Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77: 118-125.

Omernik, J.M (2004). Perspectives on the nature and definition of ecological regions. *Environmental Management* 34: S27-S38.

Omernik, J.M. and Bailey, R.G. (1997). Distinguishing between watersheds and ecoregions. *Journal of the American Water Resources Association* 33:935-949.

Orchard, A. E. (1988). A Natural Regions Map for Tasmania. Proceedings Royal Society of Tasmania 122: 47-51

Paavola, R., Muotka, T., Virtanen, R., Heino, J. and Kreivi, P. (2003). Are biological classifications of headwater streams concordant across multiple taxonomic groups? *Freshwater Biology* 48: 1912–1923.

Pease, B. (1999).A spatially oriented analysis of estuaries and their associated commercial fisheries in New South Wales, Australia. *Fisheries Research* 42: 67-87.

Phillips, B. and Butcher, R. (2006). River Parks: Building a system of 'Habitat Management Areas' across the Murray-Darling. An international and national review of freshwater 'protected areas' for conserving aquatic biodiversity and river health. Murray- Darling Basin Commission Publication No. 07/06

Pressey, R.L. (1986). Wetlands of the River Murray below Lake Hume. (Environmental Report 86/1). Murray Darling Basin Commission, Canberra

Pressey, R.L. (2004). Conservation planning and biodiversity: assembling the best data for the job. *Conservation Biology* 18:1677-1681.

Pressey, R.L. and Adam, P. (1995). A review of wetland inventory and classification in Australia. *Vegetatio* 118: 81-101.

Proches, S. (2008). Three ways to split a continent. *Journal of Biogeography* 35: 195-196.

Ramsar (2006). *Strategic Framework for the List of Wetlands of International Importance*, third edition

Roberts, J. and Butcher, R. (2006). *Wetland Classification and Condition: A background paper for the Vegetation Theme of the Sustainable Rivers Audit.* Report JR 14/2006. Canberra, ACT.

Robertson, H.A. and Fitzsimons, J.A. (2004). Hydrology or Floristics? Mapping and Classification of Wetlands in Victoria, Australia, and Implications for Conservation Planning. *Environmental Management* 34 (4): 499–507

Rosgen, D. L. (1994) A classification of natural rivers. *Catena*, 22: 169–199

Roshier, D.A., Whetton, P.H., Allan, R.J. and A. I. Robertson. (2001). Distribution and persistence of temporary wetland habitats in arid Australian in relation to climate. *Austral Ecology* 26:371–384.

Roux, D., Moor, F., Cambray, J. and Barber-James, H. (2002). Use of landscape-level river signatures in conservation planning: a South African case study. *Conservation Ecology*, 6, 6. http://www.consecol.org/vol6/iss2/art6

Roux, D.J., Nel, J.L., Ashton, P.J., Deacon, A.R., de Moor, F.C., Hardwick, D., Hill, L., Kleynhans, C.J., Maree, G.A., Moolman, J. and Scholes, R.J. (2008). Designing protected areas to conserve riverine biodiversity: Lessons from a hypothetical redesign of the Kruger National Park. *Biological Conservation* 141: 100-117.

Sandin, L., Sommerhauser, M., Stubauer, I., Hering, D. and Johnson, R. (2000). The Development and Testing of an Integrated Assessment System for the Ecological Quality of Streams and Rivers throughout Europe using Benthic Macroinvertebrates: Stream assessment methods, stream typology approaches and outlines of a European stream typology. a project under the 5th Framework Programme Energy, Environment and Sustainable Development Key Action 1: Sustainable Management and Quality of Water Contract No: EVK1-CT1999-00027.

Saunders, D.L., Meeuwig, J.J. and Vincent, A.C.J. (2002). Freshwater protected areas: strategies for conservation. *Conservation Biology* 16: 30–41.

Scott, D.A. and Jones, T.A. (1995). Classification and inventory of wetlands: A global overview. *Vegetatio* 118: 3-16.

Semeniuk, C.A. (1987). Wetlands of the Darling System - a geomorphic approach to habitat classification. *Journal Royal Society Western Australia* 69:95-112.

Semeniuk, C.A. and Semeniuk, V. (1995). A geomorphic approach to global classification for inland wetlands. *Vegetatio* 118: 103-124.

Semeniuk, V. and Semeniuk, C.A. (1997). A geomorphic approach to global classification for natural inland wetlands and rationalisation of the system used by the Ramsar Convention – a discussion. *Wetlands Ecology and Management* 5; 145-158.

Silk, N. and Ciruna, K. (2005) (eds). *A practioner's guide to freshwater biodiversity conservation*. The Nature Conservancy, Washington, DC; Island Press.

Smith, R.D., Ammann, A. Bartoldus, C. and Brinson, M.M. (1995). *An Approach for Assessing Wetland Functions Using Hydrogeomorphic Classification, Reference Wetlands, and Functional Indices*. Technical Report WRP–DE–9, U.S. Corps of Engineers, Army Engineer Waterways Experiment Station

Snelder, T.H. and Biggs B.J.F., (2002). Multiscale river environment classification for water resources management. *Journal of the American Water Resources Association* 38: 1225–1239.

Snelder, T.H. and Biggs, B.J.F. (2002) Multi-scale river environment classification for water resources management. Journal of the American Water Resources Association, 38, 1225–1239.

Snelder, T.H., Cattaneo, F., Suren, A.M. and Biggs, B.J.F. (2004). Is the River Environment Classification an improved landscape-scale classification of rivers? *Journal of the North American Benthological Society*, 23 (3): 580-598

Stein, J. L. (2005). A national landscape framework for river and stream conservation: developing a new stream and catchment reference system. Presented at *Fourth Australian Stream Management Conference: linking rivers to landscapes*. Launceston, Tasmania, 19-22 October, 2004. Rutherford, I. D., Wiszniewski, I., Askey-Doran, M. and Glazik, R. (Eds.) Department of Primary Industries, Water and Environment pp. 548-552

Stein, J.L. (2007). A continental landscape framework for systematic conservation planning for Australian rivers and streams, PhD Thesis, Centre for Resource and Environmental Studies, Australian National University, Canberra

Tait, J. T. P., Choy, S. and Lawson, R. (2003). Bioregional frameworks for assessments of freshwater biodiversity in Australia. Presented at *World Congress on Aquatic Protected Areas*. Cairns, Australia, 14-17th. August, 2002. Beumer, J. P., Grant, A. and Smith, D. C. (Eds.) Australian Society for Fish Biology pp. 155-169.

TFMPA (1999). *Understanding and applying the principles of comprehensiveness, adequacy and representativeness for the NRSMPA*, Version 3.1. Report prepared by the Action Team for the ANZECC Task Force on Marine Protected Areas. Marine Group, Environment Australia, Canberra.

Thackway, R. and Cresswell, I.D. (1995) (eds). An Interim Biogeographic Regionalisation for Australia: a framework for establishing the national system of reserves, Version 4.0. Australian Nature Conservation Agency, Canberra.

Thieme, M., Abell, R., Stiassny, M.L.J.S., and Skelton P. (2005). Freshwater ecoregions of Africa and Madagascar: A conservation assessment. Island Press.

Thieme, M., Lehner, B., Abell, R., Hamilton, S.K., Kellndorfer, J., Powell, G. and Riveros, J.C. (2007). Freshwater conservation planning in data-poor areas: an example from a remote Amazon basin (Madre de Dios River, Peru and Bolivia). *Biological Conservation* 135: 484–501.

Thomson, J. R., Taylor, M. P., Fryirs, K. A. and Brierley, G.J. (2001). A geomorphological framework for river characterization and habitat assessment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 11, 373–389.

Tiner, R.W. (1999). *Wetland indicators: a guide to wetland identification, delineation, classification, and mapping*. CRC Press, Florida, United States of America.

Turak, E. (2007). *An Ecological Typology of the Rivers of New South Wales, Australia*. PhD thesis. University of Technology Sydney, NSW, Australia.

Turak, E. and Koop, K. (2008). Multi-attribute ecological river typology for assessing ecological condition and conservation planning. *Hydrobiologia* 603:83-104.

Unmack, P.J (2001). Biogeography of Australian freshwater fishes. Journal of Biogeography 28:1053-1089.

USDA (United States Department of Agriculture) (2008). *Hydrogeomorphic Wetland Classification System: An Overview and Modification to Better Meet the Needs of the Natural Resources Conservation Service*. Technical Note No. 190–8–76. USDA, Washington.

Verdonschot, P.F.M. and Nijboer, R.C. (2004). Testing the European stream typology of the water framework directive for macroinvertebrates. *Hydrobiologia* 516: 35–54.

Ward, J.V., Malard, F. and Tockner, K. (2002) Landscape ecology: a framework for integrating pattern and process in river corridors. *Landscape Ecology.* 17 (1): 35-45.

Warner, B.G. and Rubec, C.D.A. (eds.) (1997). *The Canadian Wetland Classification System*, Second Edition. Wetlands Research Centre, University of Waterloo, Waterloo, Ontariao,

Wells, F., L. Metzeling and Newall, P. (2002). Macroinvertebrate regionalisation for use in the management of aquatic ecosystems in Victoria, Australia. Environmental Monitoring and Assessment 74: 271–294.

Whittington, J., Coysh, J., Davies, P., Dyer, F., Gawne, B., Lawrence, I., Liston, P., Norris, R., Robinson, W. and Thoms, M. (2001). *Development of a Framework for the Sustainable Rivers Audit.* A Report to the Murray–Darling Basin Commission. Technical Report 8/2001, Cooperative Research Centre for Freshwater Ecology, Canberra.

Whittaker, R.J., Araújo, M.B., Jepson, P., Ladle, R.J., Watson, J.E.M., and Willis, K.J. (2005). Conservation biogeography: assessment and prospect. Diversity and Distribution 11: 3-23.

Wikramanayake, E., Dinerstien, E., Loucks, C., Olson, D., Morrison, J., Lamoreux, J., McKnight, M. and Hedao, P. (2002). Ecoregions in ascendance: reply to Jepson and Whittaker. *Conservation Biology* 16: 238-243.

Wimmer, R., A. Chovanec, O. Moog, M. H. Fink & D. Gruber (2000): Abiotic Stream Classification as a Basis for a Surveillance Monitoring Network in Austria in Accordance with the EU Water Framework Directive. *Acta hydrochimica et hydrobiologica*: 28 (4): 177-184.