Lake Eyre Basin Rivers Monitoring Project

Hydroecology of water-dependent ecosystems of the western rivers, Lake Eyre Basin

DEWNR Technical report 2015/40

Government of South Australia
Department of Environment, Water and Natural Resources

Funding for these projects has been provided by the Australian Government through the Bioregional Assessment Programme.

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Foreword

The Department of Environment, Water and Natural Resources (DEWNR) is responsible for the management of the State's natural resources, ranging from policy leadership to on-ground delivery in consultation with government, industry and communities.

High-quality science and effective monitoring provides the foundation for the successful management of our environment and natural resources. This is achieved through undertaking appropriate research, investigations, assessments, monitoring and evaluation.

DEWNR's strong partnerships with educational and research institutions, industries, government agencies, Natural Resources Management Boards and the community ensures that there is continual capacity building across the sector, and that the best skills and expertise are used to inform decision making.

Sandy Pitcher
CHIEF EXECUTIVE
DEPARTMENT OF ENVIRONMENT, WATER AND NATURAL RESOURCES

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Funding for this project has been provided by the Australian Government through the Department of the Environment as part of the Bioregional Assessment Programme. See www.bioregionalassessments.gov.au for further information.

Contents

List of figures

List of tables

Summary

Assessments of the environmental impacts of coal seam gas (CSG) and large coal mining proposals in the Lake Eyre Basin (LEB), South Australia, are hindered by a lack of data for regional water-dependent ecosystems (WDE). Given scant data, one way to promote an objective assessment framework is to incorporate hydroecological concepts and principles into bioregional assessments[1].

This study concerns the western rivers of the Lake Eyre Basin (WRLEB), namely the Neales, Macumba and Finke catchments, which overlie coal-bearing geological formations associated with the Arckaringa and Pedirka basins. Although the region is extremely arid, the catchments support a variety of surface WDEs with many associated aquatic and terrestrial biota. Each catchment has a distinctive geomorphic character, owing to unique combinations of landforms, particularly basement outcrops, high-infiltration sand dunes and high run-off gibber plains, and these characteristics influence the distribution of waterholes and other aquatic ecosystems. The regional rivers and floodplains support woodlands, wetlands and other habitats that are drought refuges and corridors for dispersal of aquatic and terrestrial biota. These include some of the least-studied ecosystems in Australia, although knowledge of the region, particularly the Neales catchment, has increased since 2000.

This report begins with a review of concepts in arid-zone river ecology. Some, including concepts of 'boom and bust' ecology and river 'highways', have been developed partly through studies on the LEB and others, including the 'natural flow paradigm' and certain flow-ecology relationships, are applicable to rivers worldwide. Analyses of climatic, hydrological and ecological data are then undertaken to identify key 'drivers' of assemblage structure and habitats. Metrics of hydrologic variability are used to define the water regimes supporting aquatic and terrestrial species and communities, and other patch-scale variables such as water quality, soil types and elevation are employed in risk assessments for local disturbances and other activities.

The western rivers harbour 15 fish species, of which 12 species (with the alien Eastern Gambusia) are shared with the Cooper and Diamantina catchments in the east. The western fish fauna is distinguished by the dominance of hardy species and three species endemic to the Finke catchment. Five groups of indicator species are identified, associated with catchments and refuge types. Algebuckina Waterhole is confirmed as a permanent 'Ark' refuge for fish species in the Neales catchment. Relationships between the presence/absence of species and flow stages suggest that floods have a 'disorganising' influence on assemblages.

The western floodplains support a mosaic of plant species, shaped by flood history, present flow regimes and habitat conditions. Four reasonably distinct groups are distinguished, including 28 indicator species that potentially could reflect flow-regime changes. Bore-Drain Sedge and Common Reed are highlighted as two aquatic species that could become invasive under changed flow conditions.

Conceptual models of the responses of WDEs to three mining and coal seam gas (CSG) scenarios are developed from the arid-zone river ecology review and analyses, and compared with the present water regime to highlight potential indicators and responses. The models and comparisons are limited to hydrological, fish and vegetation data, representing the best available information.

Development of an adaptive planning tool will require further refinements to models, relational databases and monitoring programs in the region. The following recommendations are noted and grouped by priority:

- <u>Very high</u> (foundational activities without which risk assessment and management decisions cannot be undertaken):
    - Develop a practical framework for assessing risks to surface WDEs from flow-regime changes, including those arising from coal resource development. This should integrate and build on the results of all the LEB water knowledge projects (DEWNR 2015), the state-wide vulnerability assessment project (Berens et al 2014) and DEWNR's risk assessment framework for water planning and management (DEWNR 2012). In particular, it should be specific for the region, related to aquatic ecosystem types in the region and capable of assessing risks from specific developments. Such a framework is required for the entire LEB for the Bioregional Assessment Programme as well as for other purposes.

[1]*The Australian Government is undertaking Bioregional Assessments to elucidate the potential impacts of coal seam gas and coal mining on water resources and related assets. Refer http://www.bioregionalassessments.gov.au/*

- Review potential indicators, refine measures and identify thresholds to a standard appropriate for monitoring and detection of impacts.

- High (activities which would substantially advance understanding of the vulnerability of aquatic ecosystems and/or improve the robustness of risk assessment):
    - Extend the analyses for fish and vegetation to other biotic groups, including algae, invertebrates (micro- and macro-invertebrates), waterbirds, amphibians, reptiles and mammals, and obtain data also for microbial processes.
    - Investigate groundwater resources and surface water-groundwater interactions (Section 1.4). Research by JF Costelloe (e.g. Costelloe et al. 2005b; 2008; Costelloe 2011; Costelloe and Russell 2014) has identified groundwater interactions in waterholes in the mid-Peake and Neales catchments, but there is virtually no information elsewhere in the WRLEB. The Atlas of Groundwater Dependent Ecosystems (BOM 2014) identifies many WDEs with moderate to high potential for surface or subsurface groundwater dependence in the Neales catchment, and some in the Macumba (the Atlas does not cover the Northern Territory).
    - Document refuges and refuge types (Ark, Disco, Polo Club) for the Macumba catchment. Fish surveys undertaken for this study and LEBRA show that at least one Ark refuge is likely to exist in the Macumba catchment, but its location is yet to be determined.
    - Refine hydrological models to determine the extent and frequency of inundation associated with plant assemblages (Appendix E). The approach trialled here could be used to model the impacts of changes in flood regime on floodplain vegetation, but is limited by uncertainties in modelled stage heights above cease-to-flow levels (cf. Montazeri and Osti 2014). A priority should be to improve the accuracy of modelled stage heights.
- Moderate (activities that would substantially improve the accuracy hydroecological analysis and modelling of the WRLEB):
    - Undertake floodplain-extent mapping for the WRLEB (cf. Miles and Miles 2014). This project has made progress through remote sensing, but more work is needed in regard to maximum extent and flood ARIs. Recent mapping of open-water detection frequency by Geoscience Australia may under-estimate inundation extent and frequency, owing to short-flow durations and the small size of many waterbodies in the WRLEB.
    - Extend sap-flow monitoring, initiated in another LEB Water Knowledge project (Appendix A, Ryu et al. 2014). With analyses of xylem water and groundwater, this could be insightful in regard to water use by floodplain trees and their vulnerability to flow-regime changes. This form of monitoring should be expanded to refine indicators (Table 5.1) and assess risks from CSG and mining scenarios.
    - Investigate the detectability and abundance of species in relation to different flow stages.
    - Apply knowledge from other catchments with caution, as western catchments have distinctive assemblages. In the absence of regional data, knowledge from the better-studied eastern LEB catchments (and even the adjacent Murray-Darling Basin) may be applied to the WRLEB, but within limits. The analysis of fish data showed that there are different assemblages in eastern v. western v. Finke catchments (Section 3.4.4.1), and potentially different responses by some species to flow stages in eastern v. western catchments.

1. Introduction

1.1 Context

The Lake Eyre Basin (LEB) is one of the world's few remaining large, unregulated drainage systems (McMahon et al. 2008b), and its rivers have the most variable flow regimes in the world (Puckridge et al. 1998). Extending over 1.2 million km^2, the LEB supports a variety of landforms and streams, floodplains, woodlands and wetland ecosystems, some with nationally and internationally recognized conservation values (e.g. Coongie Lakes, artesian springs: Morton et al. 1995). In the Far North of South Australia, water-dependent ecosystems (WDEs) have immense cultural, social and ecological significance (White 2014):

> *In the South Australian arid lands water is the magnet that attracts people, biodiversity and industry. It is the key resource in an otherwise dry environment. For a region where rainfall is so low, there is an amazing ability to support a huge diversity of life through a phenomenon known as boom and bust.*

'Boom and bust' ecology (e.g. Bunn et al. 2006) is a natural consequence of summer monsoonal rainfall over the arid lands centred on Kati Thanda–Lake Eyre, the world's largest ephemeral lake (Kotwicki 1986). Many aquatic and terrestrial species in the region rely on artesian springs (Fensham et al. 2011; Davis et al. 2013), waterhole refuges and mosaics of riverine and floodplain habitats that are connected, periodically, by surface-water flows (Costelloe et al. 2003; Arthington et al. 2005; Fensham et al. 2011; Costelloe and Russell 2014). WRLEB ecosystems thereby include components and processes operating at multiple temporal and spatial scales (e.g. Davis et al. 2013).

The regional WDEs are among the least-studied Australian inland waters (LEBSAP 2008), and knowledge is needed to address increased resource development pressures anticipated over the next 5–10 years (Barrett et al. 2013). As interest grows in natural coal and coal seam gas (CSG) reserves in northern South Australia, clear industry standards based on robust scientific understanding are needed to ensure that high-value WDEs are protected (COAG 2012).

1.2 Aims and objectives

This report is part of a series of studies forming part of the Lake Eyre Basin Rivers Monitoring. Lake Eyre Basin Rivers Monitoring project is one of three water knowledge projects undertaken by the South Australian Department of Water, Environment and Natural Resources (DEWNR) to inform the Bioregional Assessment Programme in the Lake Eyre Basin. The three projects are:

- Lake Eyre Basin Rivers Monitoring,
- Arckaringa and Pedirka Groundwater Assessment and
- Lake Eyre Basin Springs Assessment (see DEWNR 2014).

The Bioregional Assessment (BA) Programme is a transparent and accessible programme of baseline assessments that increase the available science for decision making associated on potential water-related impacts of coal seam gas and large coal mining developments. A bioregional assessment is a scientific analysis of the ecology, hydrology, geology and hydrogeology of a bioregion with explicit assessment of the potential direct, indirect and cumulative impacts of coal seam gas and large coal mining development on water resources. This Programme draws on the best available scientific information and knowledge from many sources, including government, industry and regional communities, to produce bioregional assessments that are independent, scientifically robust, and relevant and meaningful at a regional scale.

Consistent with the aims of the LEB BA (Barrett et al. 2013), the aims of this study are to:

- Compile knowledge and synthesise concepts for arid zone rivers,
- Undertake conceptual modelling to understand potential impacts of altered surface flow regimes from open-cut mining and point-discharge scenarios,
- Characterize spatial and temporal patterning of climatic, hydrologic, and biotic datasets,
- Identify data gaps and improvements, and

- Consider ways to integrate hydroecological model data with assessments of mining risks and impacts.

The goal here is to present an ecological overview of the flow regimes of WRLEB rivers and the ways that these could be affected by CSG and coal mining activities. This study complements a separate LEBRM sub project (Imgraben and McNeil 2014) concerned with conceptual models related to specific types of waterbodies and mining activities.

Ultimately information and tools developed as part of this study will inform the IESC. The Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development (the IESC) is a statutory body under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) which provides scientific advice to Australian governments on the water-related impacts of coal seam gas and large coal mining development proposals.

Under the EPBC Act, the IESC has several legislative functions to:

- Provide scientific advice to the Commonwealth Environment Minister and relevant state ministers on the water-related impacts of proposed coal seam gas or large coal mining developments.
- Provide scientific advice to the Commonwealth Environment Minister on:
    - bioregional assessments being undertaken by the Australian Government, and
    - research priorities and projects commissioned by the Commonwealth Environment Minister.
- Publish and disseminate scientific information about the impacts of coal seam gas and large coal mining activities on water resources.

1.3 Approach and limitations

This report follows a 'multiple lines of evidence' approach to identify patch to landscape-scale hydroecological markers for maintenance of riparian vegetation and fish assemblages (Sections 3.4.4, 3.4.5). These two biotic groups are selected because they provide sufficient data to sustain analysis, and because they are a fair, albeit limited, representation of WDE services in the region. There are numerous other biotic groups, including algae, invertebrates, waterbirds and other vertebrates, and processes, particularly microbial processes, for which data (or expert knowledge) are sparse or missing. These omissions need to be addressed for a more complete picture, and they are recalled later among recommendations for future investigations. In particular there is potential for feedback loops to exist that may cause responses in hydrology, fish or vegetation that are not modelled in this analysis.

A further caveat is that groundwater surface-water interactions in this report are not given attention commensurate with their importance (cf. Tomlinson and Boulton 2008, 2010). Indeed, there are few linked hydrological and ecological data on ground-water-surface water interactions in arid/semi-arid wetlands (Jolly et al. 2008), and this is a major shortcoming in view of the potential environmental implications of coal seam gas and mining development. Methods are available to conduct these investigations (e.g. Brodie et al. 2007; McCallum et al. 2009), and they warrant a high priority among the obligations that the industry will need to address as development proceeds. Hydroecology and impact assessment

The discipline of hydroecology (or 'ecohydrology') connects spatial and temporal patterns in hydrology and ecology that collectively support WDEs. Evidence-based models of ecosystem components and processes are related to flow-regime characteristics to inform assessments of risk to WDEs and resource planning and policy (e.g. Green et al. 2013, 2014). Hydroecology is an inexact science, because animals and plants are influenced by other environmental and climatic drivers and may not respond directly to changes in flow. Even within the context of a flow regime, or a single modelled hydrograph, it is unrealistic to separate 'facets', like flow duration, magnitude and rate of recession, as if they might be treated independently of one another (Walker et al. 1995, Puckridge et al. 1998). Further, the responses of organisms may be linked more closely to changes in water level rather than actual flow (the two are not necessarily correlated), and the issue then is one of *hydraulics* rather than *hydrology*. The responses to changed flow regimes of animals and plants (and ecological communities) can rarely be modelled as simple cause-effect, regression-type relationships. Rather, they may be better represented by Bayesian models (e.g. Gawne et al. 2012).

In brief, hydroecology can identify measurable indicators of ecosystem functions that can be monitored against objectives, promoting a more holistic water resource policy and planning process in keeping with the principles of adaptive management (e.g. Kingsford and Biggs 2012). It may also indicate the risk of ecological objectives *not* being met, thereby reducing uncertainty in decision-making.

Methods for risk assessments that utilise only a few hydroecological relationships are likely to be less reliable than those that incorporate multiple spatial and temporal data. Ideally, resource management in the arid zone should be informed by patch- and landscape-scale processes governing water dependencies (Stafford Smith and McAllister 2008; Kennard et al. 2010; Gawne et al. 2012). Accordingly, this project seeks to characterize multi-scale relationships between flow, fish and vegetation and their environmental settings.

1.4 Concepts in arid-zone river ecology

In thinking conceptually about arid-zone rivers, it is helpful to look beyond ideas that relate specifically to rivers. For example, Morton et al. (2011) ventured 14 propositions as a framework to describe the ecology of Australian deserts, emphasizing the significance of variable rainfall and low levels of nutrients. In arid-zone ecosystems generally, the availability of water and other vital resources for animals and plants is discontinuous, occurring as discrete pulses within long periods when they are in comparatively short supply. A proper perspective requires us to adjust our frame of reference to consider processes operating on spatial and temporal scales outside our common experience (Walker et al. 1995; Stafford Smith and McAllister 2008).

Two seminal concepts in river ecology, the *River Continuum Concept* (Vannote et al. 1980) and the *Flood Pulse Concept* (Junk et al. 1989), have currency for Australian rivers in general, and dryland rivers in particular (e.g. Puckridge et al. 1998), and several more recent approaches could be explored to inform models specifically for the WRLEB. These include reviews of emerging concepts in the ecology of intermittent rivers (Larned et al. 2010, 2011; Datry 2014) and an integrated model, *The River Wave Concept* (Humphries et al. 2014), that potentially applies to all rivers governed by periodic floods.

The following text outlines several themes relevant to analyses of data for hydrology, fish and vegetation in Section 3.

1.4.1 Pulse-reserve models

One of the pivotal ideas in arid-zone plant ecology has been the ‘pulse reserve’ model formally introduced by Noy-Meir (1973, 1974). This proposed a direct link between episodes of rainfall and pulses of plant growth, providing reserves of carbon and energy to the wider ecosystem. Although the original model recognized the significance of low annual rainfall, high inter-annual variability and low predictability, later researchers (e.g. Reynolds et al. 2004) have emphasized that the plant-growth response is modified by the way that rainfall is transformed into available soil moisture and, in turn, how and when plants access the moisture. Many authors since have explored the significance of (frequent) small *versus* (rare) large events for different functional groups of plants, and a range of derivative models has been proposed.

A ‘biologically meaningful’ rainfall pulse (Reynolds et al. 2004) is more likely to follow from clusters of rainfall episodes rather than from individual events. Noy-Meir (1973) viewed the patterns of pulses and growth responses as widely spaced, clustered and highly seasonal or intermediate (discrete but temporally connected). Nano and Pavey (2013) refined these ideas with regard for plants in the Simpson Desert, confirming the significance of rainfall connectivity and soil texture (hence soil moisture), as well as seasonal growth constraints and rooting attributes. An introduction to ideas based on pulse-reserve models is provided by a workshop introduced by Schwinning et al. (2004). Indeed, the significance of connected pulses for terrestrial plants and ecosystems has a parallel in ideas of the cumulative consequences of serial floods in arid-zone rivers (e.g. Puckridge 1999; Leigh et al. 2010; Puckridge et al. 2010).

1.4.2 Telescoping Ecosystem Model

Fisher et al. (1998) proposed a hierarchical model in which subsystems within river ecosystems are nested in one another, in the manner of elements in a telescope. The subsystems are interconnected and include the stream itself, the riparian zone and the saturated sediments beneath and alongside the channel. Materials within each subsystem are chemically transformed within a characteristic ‘processing length’, akin to a cylindrical element of a telescope. The processing length is increased by disturbance, then decreased as succession proceeds to restore the natural condition. There are parallels with the ‘reset distance’ in the *Serial Discontinuity Concept* of Ward and Stanford (1983). Fisher et al. (1998) proposed that flooding (as a disturbance) causes proportionate changes in processing length, and that the elements at the end of the telescoping hierarchy, particularly the riparian zone, would show most *resistance* but least *resilience*. Thus, resistance and resilience are seen as 'inversely correlated and spatially separated'.

The *Telescoping Ecosystem Model* was refined by Sponseller et al. (2013), who observed that the roles of water (providing habitats and connectivity between habitats, promoting exchanges of energy and transport of matter and driving geomorphic changes) vary across spatial scales and interact hierarchically. The main determinant of ecological processes is the duration of water availability, and during drought in desert rivers its influence 'collapses' to include only particular reaches or segments. Understanding how changes in the spatial extent and overlap of the roles of water shape ecological patterns is a key to predicting how river ecosystems respond to disturbance.

1.4.3 Natural Flow Paradigm

The *Natural Flow Paradigm* is based on the premise that the ecological integrity of river ecosystems depends on their natural dynamic character (Poff et al. 1997). In this context, a natural flow regime is defined by the magnitude, frequency, duration, period of occurrence and variability of unregulated flows (Poff et al. 1997, 2010; Olden and Poff 2003). Puckridge et al. (1998) acknowledged Cooper Creek, followed by the Diamantina River, to have the most variable average annual discharge of any world rivers, and identified 11 flow-regime 'facets' relevant to the regional fish fauna.

Broadly, a *flow regime* is 'a long-term statistical generalisation of the hydrograph' (Puckridge et al. 1998: p. 56). To characterize the regime, a long-term hydrograph is analysed to determine magnitude, frequency, duration, timing and rate of change in flow conditions, and averages (ideally, median values: Walker et al. 1995) are used to identify groups at a given spatial scale. This provides a streamflow classification as a context for regional management and monitoring (Kennard et al. 2010) and, coupled with reach-scale studies, an opportunity to introduce ecological understanding (Kerezsy 2010).

A streamflow classification of Australian rivers indicates 10 meta-groups, including *perennial*, *intermittent*, *highly intermittent* and *extreme intermittent* regimes (Pusey et al. 2009) (Table 1-1). The LEB is in meta-group C, including rivers in arid central and western Australia with a combined length of over 1.2 million km. This meta-group is characterised by high flow intermittency, variable runoff, wide temperature ranges and the absence of perennial and intermittent streams. It is quite distinct from the high winter rainfall, perennial streams of cool temperate southern Australia and the high summer rainfall streams of tropical northern Australia.

Streamflow classifications rely on flow gauge data, but gauges are sparsely distributed in the LEB (Table 1-1). In 2000, the ARIDFLO project (Costelloe et al. 2007) established a network of stage (water level) loggers, since maintained and modified by subsequent projects (LEB Rivers Assessment: Costelloe 2008, Cockayne et al. 2012; Critical Refugia: Costelloe 2011). Until recently, this network has provided the only flow data available for the WRLEB.

Table 1-1: Numbers of gauging stations by flow-regime classes and meta-groups (Pusey et al. 2009).

Meta-group	Flow regime class				Total gauges
	Perennial	Intermittent	Highly Intermittent	Extreme Intermittent	
A	28	6	56	4	94
B	6	1	41	26	74
C	0	0	18	22	40
D	0	26	64	18	108
E	31	10	12	0	53
F	32	57	25	1	115
G	15	53	19	0	87
H	64	87	34	2	187
I	57	4	0	0	61
J	6	0	0	0	6

Hydrological studies in the Neales catchment show that flow regimes in the WRLEB differ from those in the larger eastern catchments (Costelloe et al. 2003; Costelloe 2011). With the exception of local flows from the Innamincka Dome, the eastern-catchment flows (Cooper, Georgina-Diamantina) are characterised by monsoonal floods of varying magnitude in the upper catchments, flowing slowly through multiple channels and across broad floodplains. The downstream extent of these flows depends on prior conditions and the size of the flood, but they reach Coongie Lake and Goyders Lagoon in South Australia almost annually (Costelloe 2008, 2013). In contrast, localized convective thunderstorms drive smaller flows in the upper and lower reaches of the WRLEB (Costelloe et al. 2005a; Costelloe 2008, 2011). Catchment-wide flood events tend to be of shorter duration than those in eastern catchments, resulting in shorter periods of connectivity between waterholes and less-frequent periods of floodplain inundation. Diffuse discharges from GAB springs in near-channel or floodplain areas also interact strongly with spatial and temporal patterns of river salt balance (Costelloe et al. 2005b) (Figure 1-1).

River flows are influenced by the runoff characteristics of water-shedding landforms. For example, soils and surface hardness vary markedly between gibber mesas or plains (e.g. Neales River) and extensive aeolian or colluvial sands and gravels (e.g. Arckaringa Creek, Stevenson and Finke Rivers), resulting in variable runoff across catchments. Even so, the size and extent of rainfall events generally are good predictors of flow-pulse magnitude (Montazeri and Osti 2014).

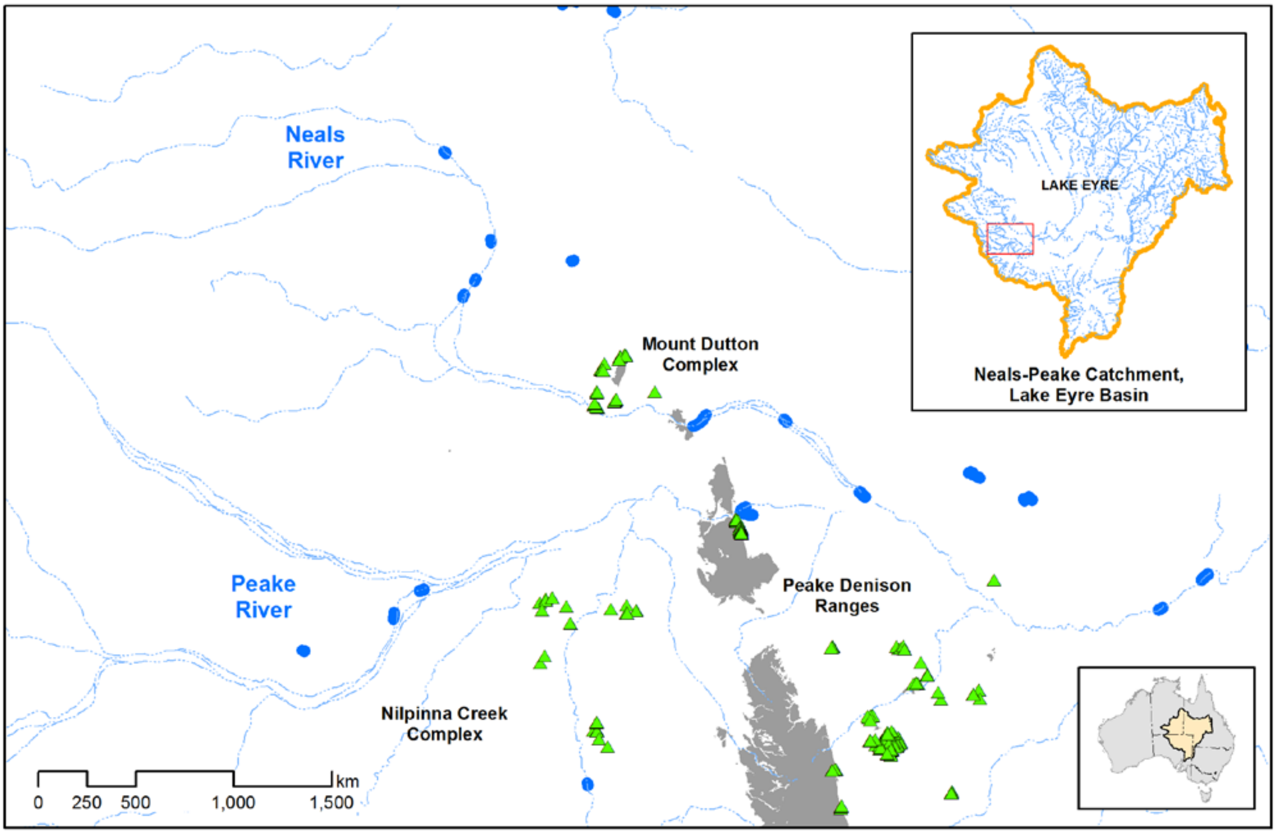


Figure 1-1: Neales–Peake catchment locations where salt from GAB springs (Mt Dutton, Nilpinna) accumulates in floodplain sediments. After Costelloe et al. (2005b)

1.4.4 Flow-ecology relationships

Flow-ecology research seeks to characterize ecological functions driven by particular facets of the flow regime. It will be necessary to determine empirical relationships for the WRLEB (Arthington et al. 2006) to properly assess risks from mining and CSG development (LEBSAP 2008). Without this information, it is difficult to specify which descriptors of flow variability are ecologically most meaningful (Olden and Poff 2003), and which elements of the natural flow regime should be retained or restored to maintain ecological integrity (Poff et al. 2010). Arid-zone flow and no-flow regimes inherently are difficult to classify as they cover a very wide range of spatial and temporal scales, but models may be less accurate if regimes and habitats are defined merely on an event-to-event and location-to-location basis (Knighton and Nanson 2001; Kennard et al. 2007).

Several studies, mainly in the eastern LEB, have shown that flow variability drives wide fluctuations in fish, bird, invertebrate and plant populations (e.g. Arthington et al. 2005; Capon 2005; Shiel et al. 2006; Costelloe et al. 2007; Kennard et al. 2007; Kingsford et al. 2014). Large species- and community-shifts may follow different events, from short pulses of high magnitude to sequential pulses of low magnitude, influencing the dispersal and recruitment of populations. Variable periods of connection and disconnection between refuge habitats also are part of the natural regime, influencing the composition and trophic structure of ecological communities (Sheldon et al. 2010).

Flow-ecology relationships in arid-zone rivers may be explored using multivariate methods, relating environmental and hydrological characteristics to 'assemblage structures'. For example, Arthington et al. (2005) compared the fish assemblages of Cooper Creek floodplains with regard for temporal changes in waterholes and river reaches and habitat variables. They reported a general decline in diversity, and a 93% decrease in abundance as waterholes contracted after less than six months of drought (April–September 2001). Reach-scale indicators of habitat, such as floodplain width and distance to nearest waterhole, proved to be effective predictors of drought-surviving populations.

Desert fish endure wide population fluctuations by rapidly colonizing new habitats ('opportunists') or spreading their reproductive effort ('periodic strategists') (Arthington and Balcombe 2011). During periods of flow, the fish move to floodplain and waterhole habitats where they build energy reserves for breeding, recruitment and survival (Kerezsy et al. 2014), although this may render them vulnerable to long dry spells. For example, during the 2001 drought in Cooper Creek, populations of Bony Herring and Spangled Grunter, two of the first species likely to re-colonise after drought, declined by 83-85% (McNeil and Schmarr 2009). Most of the fish species have strong dispersal instincts and are quick to capitalise on favourable flows, but they are also tolerant of rising salinity (often exceeding seawater), low dissolved oxygen (a consequence of salinity) and loss of microhabitats in isolated waterbodies. Their responses are also influenced by the availability of food—the hatching success of zooplankton propagules in floodplain sediments varies with the time since last flooding (Jenkins and Boulton 2003, 2007).

In contrast, riparian vegetation assemblages are likely to be structured on longer time scales, with slow advances and retreats of dominant, long-lived species in response to shifts in flow regimes or climate. Over short time frames, subtle changes in plant vigour may attenuate water imbalance and reduce seed-crops or decrease resistance to insect herbivores and disease (Manion 1981). The ephemeral components of floodplain vegetation (annual grasses and forbs) are likely to persist in soil seedbanks and remain fairly resistant to drought (Capon and Brock 2006) and altered flow regimes.

1.4.5 Boom and bust ecology

In arid-zone river ecosystems, 'booms' follow floods and 'busts' occur during drought. In central Australia, south-tending subtropical low pressure systems may bring basin-wide floods and a boom to sustain rangelands and wetlands for several years of drought (Kingsford and Porter 1999; Bunn et al. 2006; Costelloe et al. 2007; Stafford Smith and McAllister 2008; McIlwee et al. 2013). In the 20th century, there were major rainfall episodes in 1949–52, 1974–77 and 1984–85, filling Kati Thanda–Lake Eyre (Kingsford and Porter 1999; Williams 1990). Typically, the booms in productivity do not last more than a few years, and large tracts of the basin often are in drought, with evaporation exceeding rainfall (McMahon et al. 2005). In bust conditions, the productive areas contract to green corridors along water courses, and water-dependent biota must rely on dormancy (e.g. resting stages) or refuges for survival. In extended drought, the refuge habitats become critical, especially for obligate aquatic biota such as fish (Sheldon et al. 2010; Arthington and Balcombe 2011). The refuges often are waterholes that enable survival and provide colonists when the drought is ended (Robson et al. 2008).

The spatial variability of productive patches is an important feature of terrestrial arid-zone ecosystems. Microrelief along rivers and floodplains creates run-on/run-off dynamics that distribute water and nutrients into a habitat patchwork or 'mosaic' (Pickup 1991; Ludwig et al. 1996). These mosaics are used by desert animals moving through the landscape. Some rodents, for example, have extraordinary fecundity, reaching 'plague' proportions in wet seasons (e.g. Plains Rat *Pseudomys australis*; Desert Mouse *P. desertor*), but they require refuges during drought (Brandle et al. 1999; Read et al. 1999). Other species are able to access alternative resource 'windows' (areas that are productive at different times).

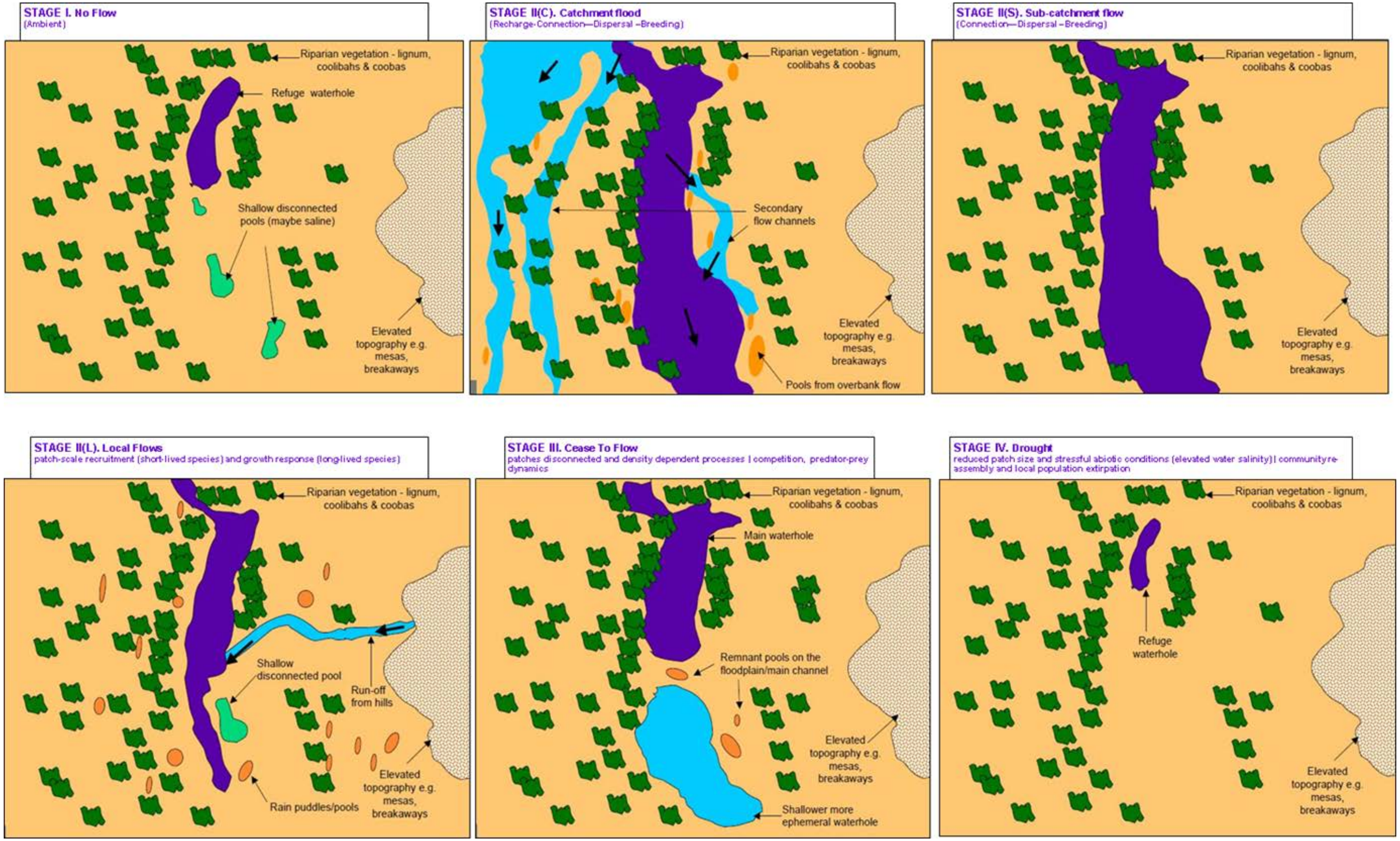


Figure 1-2: ARIDFLO conceptual model, showing four stages in the flow dynamics of LEB rivers. After Costelloe et al. (2007)

1.4.6 The ARIDFLO model

ARIDFLO, a four-year (2000–03) project in the LEB (Costelloe et al. 2003, 2007), sampled habitats for fish, macro- and micro-invertebrates and algae and recorded physical, chemical and hydrological conditions. The project involved biennial visits to three catchments (Neales, Cooper, Georgina-Diamantina), and coincided with consecutive flow pulses (2000-01), a drought (2001-02) and another pulse (2003). The ARIDFLO model incorporates elements of the Natural Flow Paradigm and flow-ecology relationships in the context of 'boom and bust' ecology. Four flow stages are recognized (Figure 1-2):

- The baseline (**Stage I: No Flow**) favours isolated permanent and semi-permanent wetlands and drought-resistant, perennial floodplain vegetation. Riparian vegetation is dominated by drought-tolerant perennials such as Gidgee (*Acacia cambagei*) and Coolabah (*Eucalyptus coolabah*). The conditions may change with rainfall from local convective thunderstorms at any time of year, or from subtropical low-pressure systems in summer (McMahon et al. 2005).
- Flows (**Stage II: Flow**) are characterized by spatial scales. Localized rainfall drives local flows (Stage II_L), whereas widespread summer rainfall drives sub-catchment flows (Stage II_S) or catchment floods (Stage II_C). Local and sub-catchment flows are confined to stream channels and connect riverine wetlands and recharge bank storage, stimulating the growth of riparian vegetation. Catchment floods inundate floodplains and connect entire catchments, resulting in productivity 'booms', creation of new habitat and dispersal of mobile species. Under high-flow conditions, strong dispersers (e.g. Spangled Grunter, *Leiopotherapon unicolor*) and flow-responders (e.g. Golden Perch, *Macquaria ambigua*) are free to move over extensive areas (Cockayne et al. 2015).
- Flow recessions (**Stage III: Cease to Flow**) typically occur over months, depending on the magnitude of flow, and the network contracts progressively to disconnected pools and reaches. Populations decline, short-lived species are lost and perennial species are exposed to competition as resources are depleted (Arthington and Balcombe 2011).
- The effects of dry conditions (**Stage IV: Drought**) vary with the spatial and temporal scale. Droughts of more than two years' duration often occur and cause drying of 'semi-permanent' wetlands. 'Permanent' waterholes are deeper (>4 m: Costelloe 2010) and surrounded by high-runoff surfaces (e.g. gibber plains and breakaways) that deliver more frequent local flows. In extended drought, there are gradual shifts in patch-interpatch boundaries on the floodplain, often with loss of vegetation in outlying areas (Figure 1-3).

Figure 1-3: Coolabah mortality at Stewarts Waterhole, Neales River (February 2014)

1.4.7 Highways for connectivity and dispersal

Many aquatic animals in desert rivers are able to spread and colonise rapidly over vast areas during periods of flow (e.g. Kerezsy 2010). Arid rivers are like 'highways' along which fish travel, pausing to feed and breed, and most species move between many 'rest stops' in their lifetimes. Connections within and between channels and wetlands are predictors of the composition of fish assemblages in waterholes (Arthington et al. 2005) and floodplain wetlands (Humphries et al. 1999; Balcombe et al. 2007), although the WRLEB wetlands may be less important for fish than those in the eastern catchments (Costelloe et al. 2007). Other species decouple their reproductive effort from flow and remain in disconnected, permanent refuge habitats; these include 'equilibrium' species, such as large-bodied fish that are 'nesters' (e.g. Cooper Catfish, *Neosiluroides cooperensis*). Thus, mixed assemblages are the norm—they confer diversity, population stability and links to biophysical processes, such as biomass and nutrient transfers, that drive productivity (Sheldon et al. 2010; Arthington and Balcombe 2011).

Localised extinctions of fish and other aquatic species may occur in sub-catchments without permanent refuges, and re-colonization then occurs during catchment-scale floods that restore connections to source populations (Kerezsy et al. 2013). The Finke catchment is a special case, as it is disconnected from other catchments under current climatic conditions (it is also home to three endemic fish species: Table 2-1; Unmack 2001, 2013).

1.4.8 Synthesis of concepts

Desert river ecosystems are shaped by variable flows that govern the distribution and movements of species and the diversity, production and the resilience of populations (Kingsford et al. 2014). The native flora and fauna are attuned to their capricious environment, and are likely to be adversely affected if flows are regulated. Indeed, desert streams may be the most sensitive of all Australian rivers to water resource development (Walker et al. 1997; Arthington and Balcombe 2011). Assessments of CSG and coal mining impacts will require better understanding of flow-ecology relationships in the LEB, using metrics and other tools to characterise flow regimes (Poff et al. 2010) in combination with ecological data to enable prediction of environmental implications. To minimise environmental impacts of development, management priorities need to include maintenance of connectivity and ecosystem integrity through protection of refuge habitats (Sheldon et al. 2010; Arthington and Balcombe 2011; McNeil et al. 2013).

Most prior studies relevant to mining-impact assessment relate to the eastern rivers of the basin, but this kind of information should be extrapolated across catchments only with caution. The western rivers are likely to function differently, given more intense aridity, ephemeral flows, higher salinities, smaller waterbodies, different species assemblages and fewer permanent refuges (Costelloe et al. 2004; Costelloe and Russell 2014). The importance of the differences between eastern and western rivers is supported by recent work for the Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development (Auricht Projects 2014, 2015) and the results of this study.

2. Regional setting

2.1 Water, land use and culture

Kati Thanda–Lake Eyre is a drainage terminus for one sixth of continental Australia, extending over arid and semi-arid bioclimatic zones and four States (Figure 2-1). The extensive north-eastern catchments, the Georgina-Diamantina rivers and Cooper Creek, carry flows from central Queensland and the Northern Territory, with occasional intense monsoonal rainfall that feeds anastomosing channels in the mid-reaches (the 'Channel Country'), where there are pastures and wetlands (e.g. Coongie Lakes, Goyder Lagoon) that support migratory birds and other aquatic biota (Figure 2-2) (Balcombe et al. 2007; Reid et al. 2009; Arthington and Balcombe 2011; Reid and Gillen 2013). In comparison, the WRLEB catchments (Finke, Macumba, Neales) are comparatively small, with narrow floodplains, and the regional rainfall is more erratic. Streams flowing to Kati Thanda–Lake Eyre South (e.g. Margaret Creek) and Lake Frome receive temperate winter and subtropical summer rainfall (McMahon et al. 2005).

The ecology and culture of humans in arid lands are underpinned by water (SAALNRMB 2010). In the WRLEB, the traditional owners form three language groups: the Arabana people of the western lowlands, the Antakarinja of the stony country and the Arrernte of the desert (Measham and Brake 2009). Water places and their ecological significance are themes for many Dreaming songs and stories that convey knowledge and a sense of connection to the land (White 2014). From the early 20th Century, water has been a catalyst for development of trade routes and the pastoral industry, and today there are about 2000 residents in towns and pastoral stations across the WRLEB. Pastoral tenure and protected estates (conservation and traditional habitation) are the main land uses (Figure 2-7), and mining, pastoralism and tourism are the dominant industries.

Historically, permanent natural springs in the WRLEB have been prized resources for graziers and traders. The springs maintain a steady flow, providing permanent, often very shallow fresh waters. Most of them issue from fracture lines deep in the Great Artesian Basin (GAB), a vast groundwater basin that underlies most of the LEB (Figure 2-6). Although the GAB has more than 2000 times the storage of the Murray-Darling Basin (about 64 million GL: Measham and Brake 2009), recharge in the western GAB effectively is zero (NWC 2013). In 1878, pastoralists drilled the first artesian bore, and their water needs since have been supplied mainly by bores. Historically, bores have been left to discharge freely, with about 95% of flow lost to evaporation and seepage (NWC 2013). Discharge rates have reduced for many bores, and many have been capped, including all of those in South Australia. Groundwater extraction also has reduced flows in natural springs, and some have ceased to flow (Fensham et al. 2010). Managing development pressure on springs and waterholes is a priority for maintaining the integrity of WDEs in the South Australian arid lands (SAALNRMB 2010, 2014).

Stock densities in the arid lands generally are low, but in dry periods the productive areas contract to riverine habitats and floodplains and grazing pressure there is intensified (Gillam and Urban 2013). In times of drought, green feed is highly valued and shrubs and even small trees are grazed. Overgrazing has led to declines of some species (e.g. Bladder Saltbush, *Atriplex vesicaria*) and changed plant community composition and recruitment, particularly in areas near water (Landsberg et al. 2003).

Permian coal beds, other hydrocarbon sources and more localized mineral deposits (gold, copper, uranium) occur in the WRLEB, as in the eastern basin (SAALNRMB 2010; Wohling et al. 2013; Keppel et al. 2014). Interest in the hydrocarbon reserves has been limited, but given the ecological and cultural values of water-dependent assets in the region, the prospect of future development raises concerns about possible adverse effects on the environment and human communities (Kingsford et al. 2014). The natural springs are vulnerable to changes in aquifer pressure (Green et al. 2013), and paradoxically the waterhole refuges and other surface WDEs—although subject to a variable natural regime—are also vulnerable to changes in the pattern of flows (Walker et al. 1997; Arthington and Balcombe 2011; Costelloe and Russell 2014).

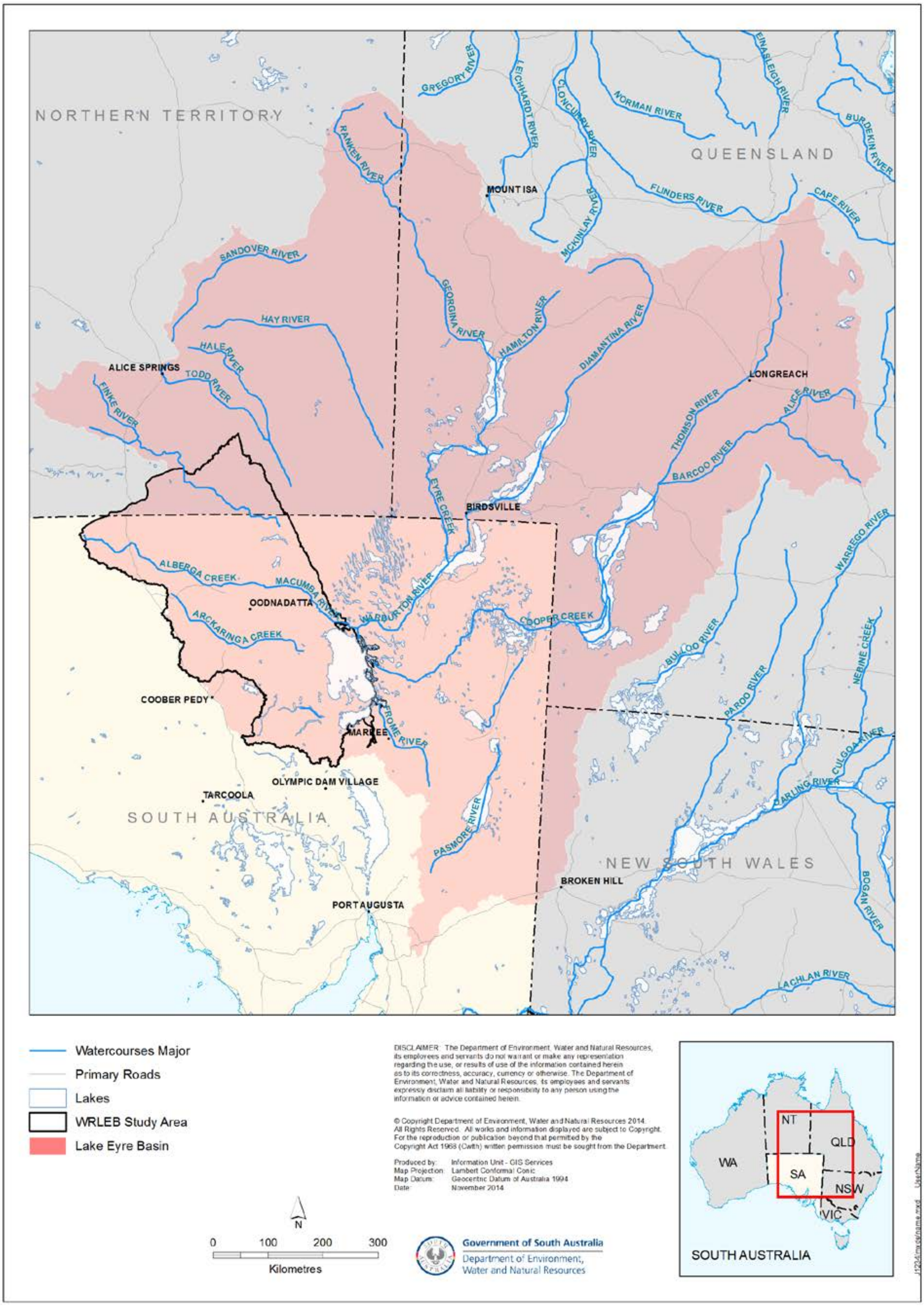


Figure 2-1: Features of the Lake Eyre Basin

Figure 2-2: Channel Country of the Diamantina River, Queensland (photo: G Scholz)

2.2 Distribution and status of water-dependent ecosystems

The ephemerality and salinity of surface WDEs in the WRLEB have limited water resource development, the hydrology of most systems is little changed and the catchments have been assessed as in 'good' condition despite scant data (LEBSAP 2008). Some terrestrial species and ecological communities associated with floodplains and flood-out environments in the WRLEB are listed under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*, but no aquatic species or wetland communities are listed as threatened (Gillam and Urban 2013). The region nevertheless does have major significance for conservation (Costelloe and Russell 2014).

The Neales catchment has received the most thorough assessment and monitoring of aquatic biota, through the ARIDFLO (Costelloe et al. 2007) and Critical Refugia projects (McNeil and Schmarr 2009) and the LEB Rivers Assessment (LEBRA) (Cockayne et al. 2012). Some sites in the Macumba and Finke catchments have been monitored by LEBRA since autumn 2011 (Cockayne et al. 2012), and one Macumba site has been monitored sporadically by the South Australian Environment Protection Authority (EPA 2008). There have also been one-off biological surveys in South Australia (Arid Rivers Biological Survey[2]) and the Northern Territory (Eldridge and Reid 1998; Duguid et al. 2005), but the lower Macumba and Finke rivers remain the least-studied ecosystems in the WRLEB.

Waterholes are likely to be refuges for aquatic and terrestrial water-dependent biota (Davis et al. 2013). Three classes of refuge are recognized (Robson et al. 2008; see also McNeil et al. 2011):

- *Ark refuges*: permanent habitats that potentially support all aquatic species in the catchment,

[2] *Department of Environment and Heritage, South Australia: unpublished data, 2005–06*

- *Polo Club refuges*: permanent and near-permanent (often saline) habitats suitable for only a few species, and
- *Disco refuges*: non-permanent habitats that are inundated for long periods in wet seasons but dry during drought.

Waterholes are located mainly along the mid- to lower reaches of the western river catchments (Figure 2-8). Larger water-bodies occur along Allapalilla Creek (Macumba), the lower Finke River and around the Dalhousie Springs complex, but these wetlands are little known and there may be only anecdotal evidence of their biota.

Water across a catchment is a spatial and temporal mosaic of potential refuge habitats, as there is little chance that all waterholes in a drainage network would be dry at one time. Waterholes in the WRLEB are reliant on episodic river flows (Costelloe 2011), but the significance of near-surface or alluvial groundwater generally is unclear. Riparian woodlands and shrublands in some areas may depend on surface-groundwater interactions (Bureau of Meteorology 2014), including parts of the Arckaringa and Peake creeks and the lower Neales and Macumba rivers. There may be groundwater-dependent reaches in Quaternary alluvial sands (light-brown regions: Figure 2-9, left panel), underlain by Cretaceous formations (Bulldog Shale) (green regions: Figure 2-9) that function as an aquitard. There are springs along the Peake-Dennison Ranges (Neales–Peake catchment) and in the Dalhousie Springs complex. Spring complexes occur also in Kati Thanda–Lake Eyre (north and south), so that the lake itself is influenced by surface-groundwater interactions.

2.3 Overview of water-dependent ecosystems

2.3.1 Neales catchment

Water in the Neales catchment is ephemeral, with only one known potentially permanent, fresh waterhole, Algebuckina (Figure 2-3). Of 19 studied waterholes, most (including Algebuckina) are surface water-dependent and three receive inputs from alluvial groundwater (Costelloe and Russell 2014; Montazeri and Osti 2014).

Ten native and one alien fish species (Eastern Gambusia, *Gambusia holbrooki*) occur in the Neales catchment (Table 2-1), although the status of Golden Perch is uncertain following surveys in 2013–14 (Schmarr et al. 2014). The assemblage includes a subset of species in the eastern catchments, but the genetic linkages between populations are unknown. During extended drought, Algebuckina Waterhole supports the entire fish assemblage of the Neales catchment and is thereby an Ark refuge (McNeil and Schmarr 2009; Costelloe and Russell 2014) (Figure 2-3). In addition, shallow, spring-fed pools in Peake Creek are likely to be permanent refuges for small fish (McNeil et al. 2009), and stable, shallow bores and springs are habitats for Eastern Gambusia. Some farm dams on outer floodplains support assemblages typical of Disco refuges (e.g. Bony Herring, *Nematalosa erebi*; Spangled Grunter; Desert Rainbowfish, *Melanotaenia splendida tatei*: McNeil et al. 2011; Cockayne et al. 2012).

In the Neales catchment, the diversity (44 species) and abundance of waterbirds were the lowest of all LEB reaches monitored by ARIDFLO, due probably to the relatively small area of suitable habitats. Floods from western catchments may contribute to filling of Kati Thanda–Lake Eyre, supporting breeding by migratory waterbirds (Costelloe et al. 2004).

2.3.2 Macumba catchment

The Macumba River discharges to Kallakoopah Creek, an anabranch of Warburton Creek in the lower Georgina-Diamantina catchment. There are many semi-permanent waterholes (Figure 2-4) and no known permanent natural waterholes, but possibly a permanent bore-fed wetland at the junction of Stevenson and Hamilton creeks (Figure 2-5). Nine native fish species occur (Cockayne et al. 2012), all shared with the Georgina-Diamantina system. The nearest known permanent waterhole to the Macumba River–Kallakoopah Creek confluence is Pandie Pandie Waterhole in Goyders Lagoon, 300-400 km upstream.

Figure 2-3: Algebuckina Waterhole at sunset. The only known permanent waterhole in the WRLEB, and an Ark refuge for fish in the Neales catchment

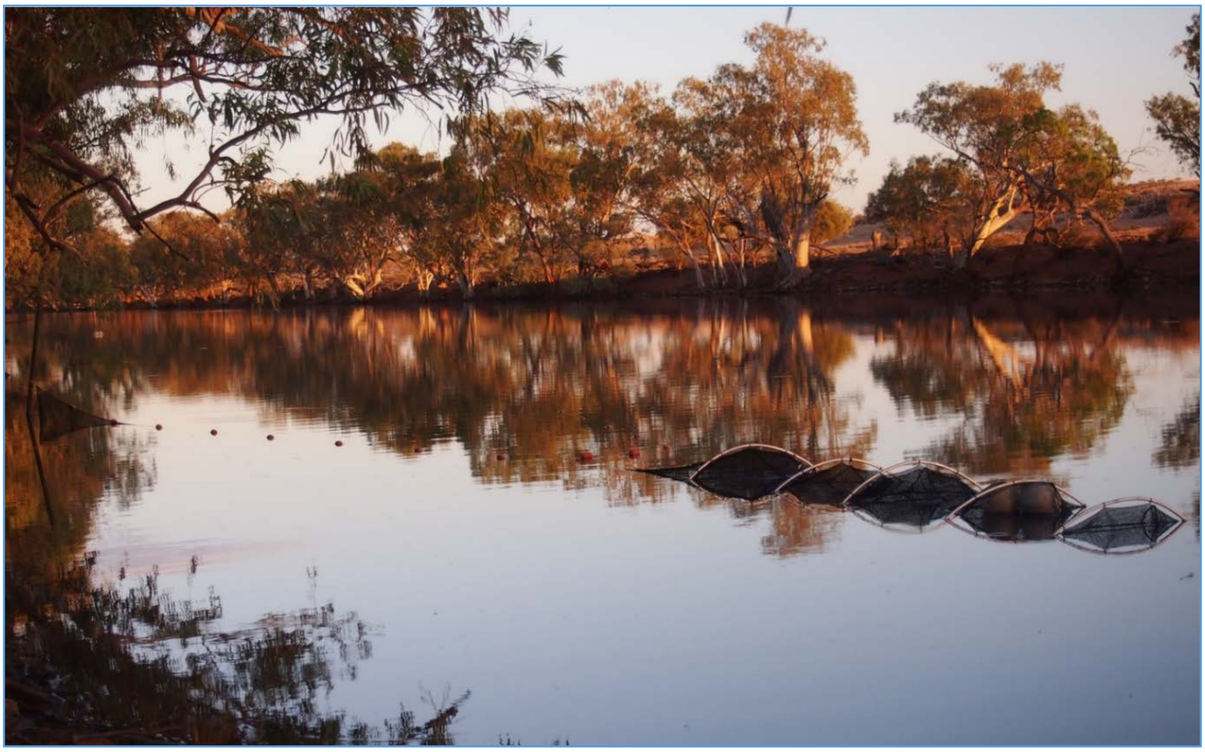


Figure 2-4: Eringa waterholes, near the Lindsay River headwaters (Macumba catchment)

Figure 2-5: Junction bore wetland at the branch of Stevenson and Hamilton creeks (Macumba catchment) after a dry summer, March 2014

2.3.3 Finke catchment

North of the Macumba catchment, the Finke River flows into the Simpson Desert dunefields. The river does not flow to Kati Thanda–Lake Eyre under present climatic conditions, but may have flowed to the Macumba catchment between 1000 and 20,000 years ago (Unmack 2001) and during more recent, occasional 'mega floods' (Pickup 1991). The Finke catchment includes numerous permanent, seasonal and ephemeral waterholes, and some of the woodlands along channels, rivers and swamps receive alluvial groundwater inputs, but the permanent waterholes are upstream of the study area (Duguid 2011). Large, rarely filled inter-dune lakes and pans and wooded swamps in the Finke floodouts are a drought refuge for terrestrial biota (Eldridge and Reid 1998). Nine native fish species occur, including three endemics (Table 2-1). No surveys have been undertaken within the study area, but fish and piscivorous waterbirds do occur in the Finke floodout (Duguid 2011).

2.3.4 Other catchments

Aquatic ecosystems in other WRLEB catchments are poorly documented. Three species of fish have been observed at road crossings in Margaret Creek and Stuarts Creek (Table 2-1), suggesting that permanent refugia exist there, probably supported by GAB springs. There are also many wetlands in claypans between dunes, filled by rainfall rather than river flows.

Table 2-1: Fish species in the Lake Eyre Basin. After Cockayne et al. (2012), with data from this study

Species	Abbr.	Common name	Cooper	Diamantina	Finke	Margaret, Stuarts	Macumba	Neales
Ambassis mulleri	Amb.mul	Desert Glass Fish	+	+	+		+	
Amniataba percoides	Amn.per	Barred Grunter		+	+		+	+
Bidyanus welchi	Bid.wel	Welch's Grunter	+	+			+	+
Bidyanus bidyanus	Bid.biy	Silver Perch		+				
*Carassius auratus**	Car.aur	Goldfish	+					
Chlamydogobius eremius	Chl.ere	Desert Goby		+		+		+
Chlamydogobius japalpa	Chl.jap	Finke Goby			+			
Craterocephalus centralis	Cra.cen	Finke Hardyhead			+			
Craterocephalus eyresii	Cra.eyr	Lake Eyre Hardyhead	+	+		+	+	+
*Gambusia holbrooki**	Gam.hol	Eastern Gambusia	+	+				+
Glossogobius aureus		Golden Goby		+				
Hypseleotris spp.	Hyp.spp	Carp Gudgeon species complex	+					
Leiopotherapon unicolor	Lei.uni	Spangled Grunter	+	+	+	+	+	+
Macquaria ambigua		Murray-Darling Golden Perch		+				
Macquaria ambigua	Mac.amb	Lake Eyre Golden Perch	+	+			+	+
*Maccullochella peelii***		Murray Cod	+					
Melanotaenia splendida tatei	Mel.spl	Desert Rainbowfish	+	+	+		+	+
Mogurnda larapintae	Mog.lar	Finke Purple-Spotted Gudgeon			+			
Mogurnda clivicola		Flinders Ranges Mogurnda	+					
Mogurnda sp.		Frew Mogurnda		+				
Nematalosa erebi	Nem.ere	Bony Herring	+	+	+		+	+
Neosiluroides cooperensis	Neo.coo	Cooper Catfish	+					
Neosilurus hyrtlii	Neo.hyr	Hyrtl's Catfish	+	+	+		+	(+)
*Oxyeleotris lineolatus***		Sleepy Cod		+				
Porochilus argenteus	Por.arg	Silver Tandan	+	+			+	(+)
Retropinna semoni	Ret.sem	Australian Smelt	+					
Scortum barcoo	Sco.bar	Barcoo Grunter	+	+				
Totals			**17**	**18**	**9**	**3**	**10**	**9**

* Introduced non-Australian species ** Introduced Australian species † Survey sites and dates in Appendix A (+) Prior to 2007 only

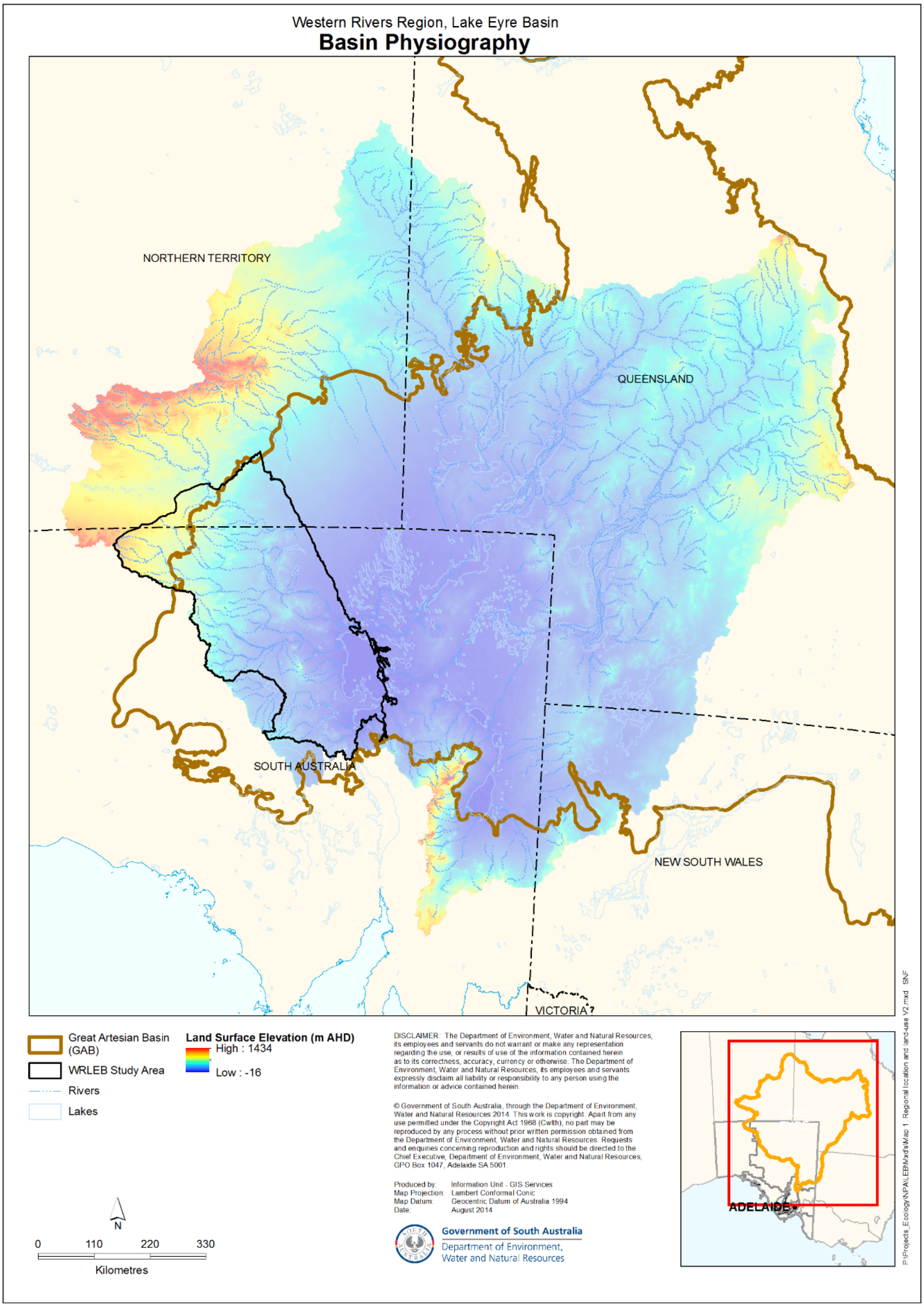


Figure 2-6: Lake Eyre Basin physiography, showing the location of the WRLEB

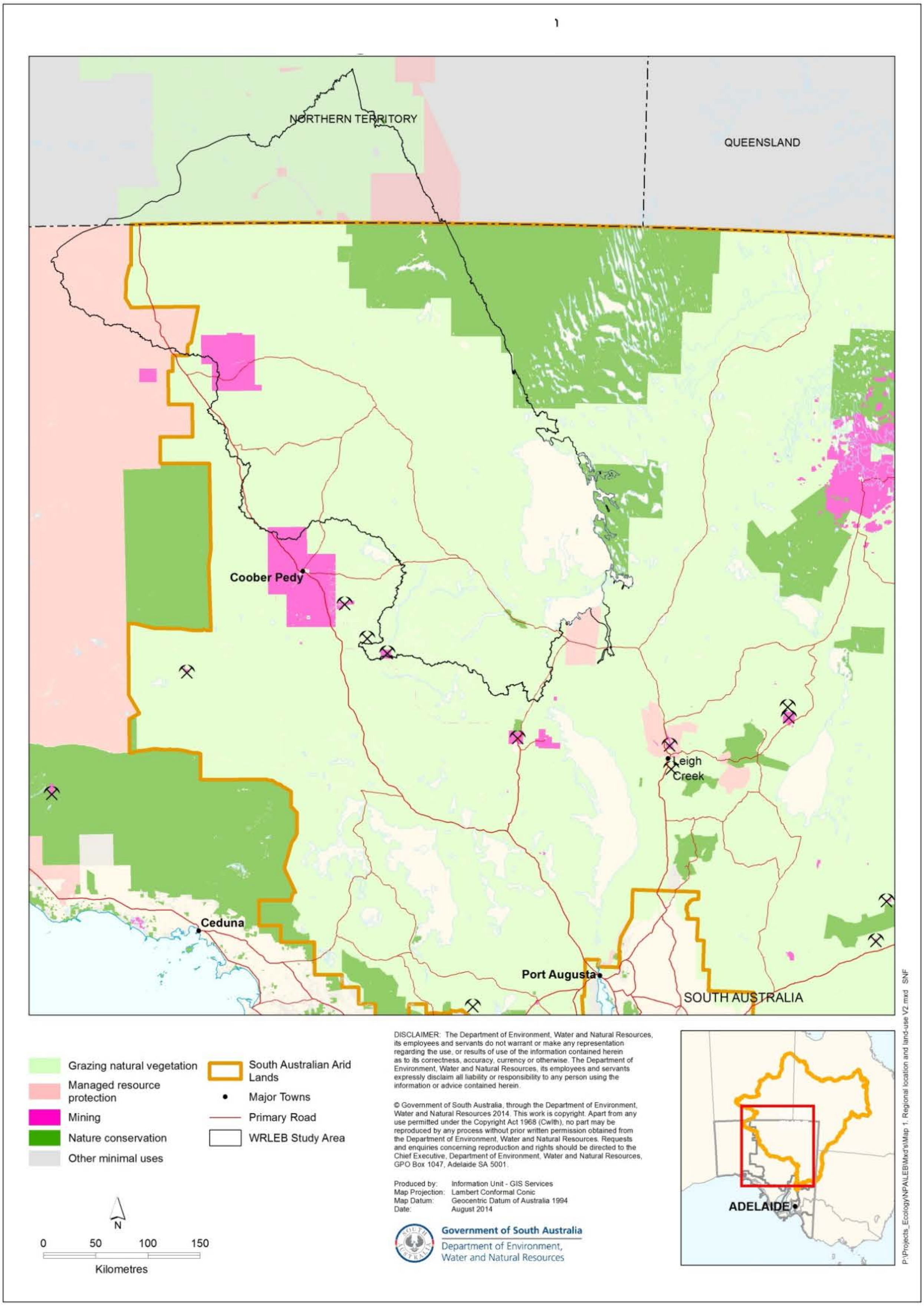


Figure 2-7: Land use in the WRLEB

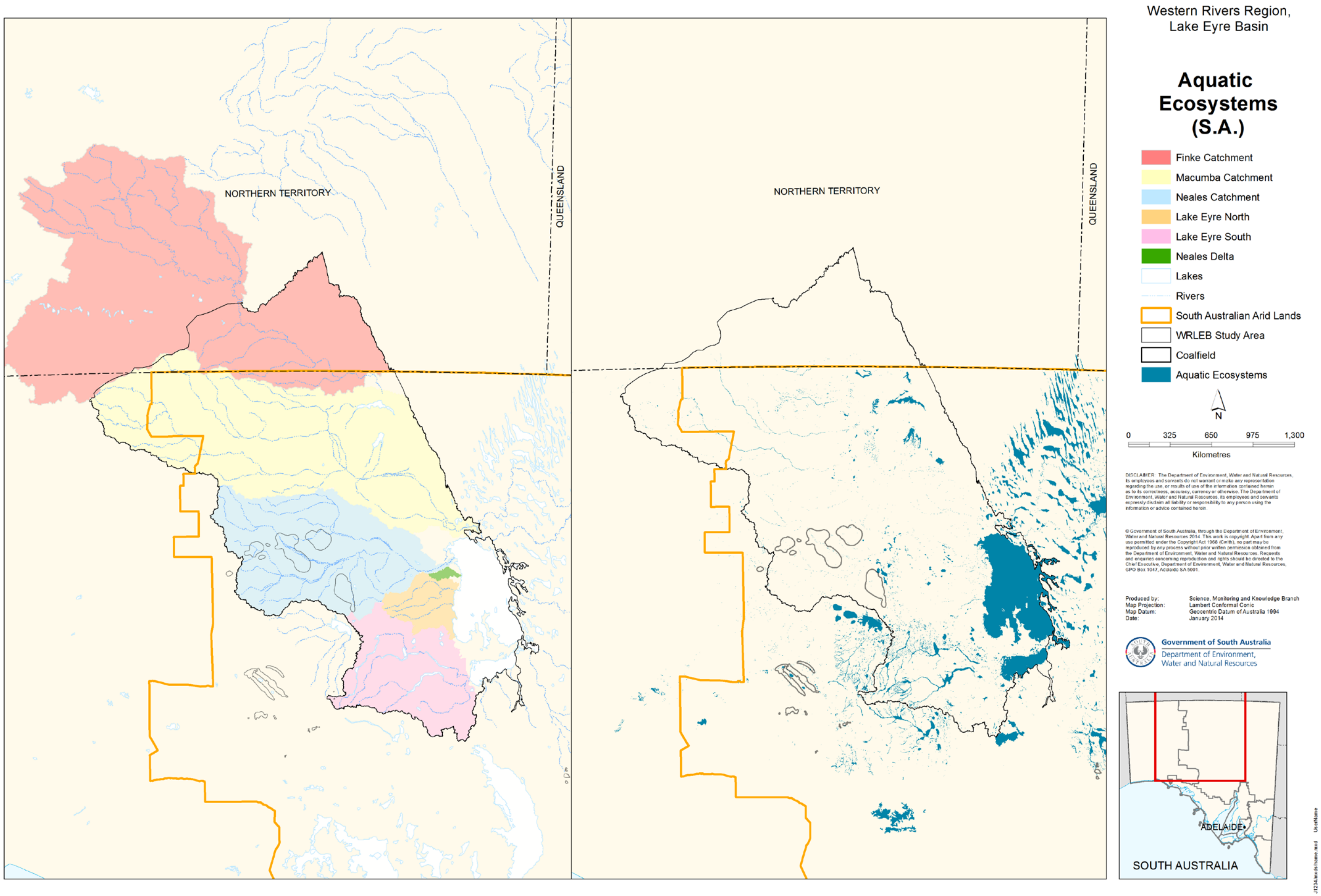


Figure 2-8: Surface catchments (left panel) and aquatic ecosystems (right panel) of the WRLEB (after Miles and Miles 2014)

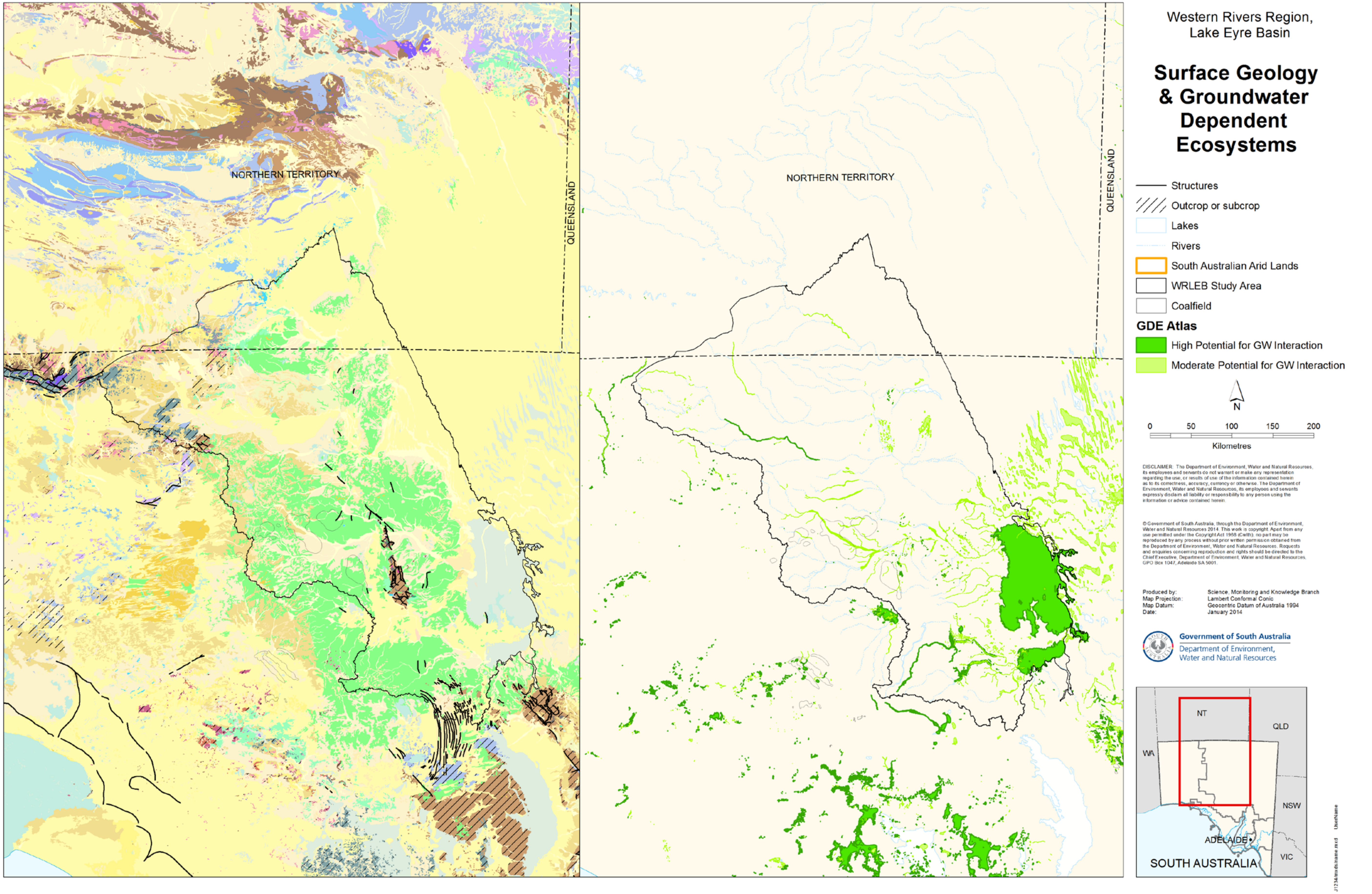


Figure 2-9: Surface geology (left panel) and surface expression groundwater-dependent ecosystems (right panel) of the WRLEB

2.4 Geomorphic setting

The main catchments of the WRLEB (Finke, Macumba, Neales) support a variety of water-shedding landforms (Figure 2-10). Each is dominated by gibber-gilgai plains occupying 42-57% of the area (Table 2-2). The plains have a gently undulating surface and a near-complete stone mantle over clay loam soils. The mantle is hard and thin (0-20 cm), impervious to rainfall and resistant to erosion. If it is disturbed, by mining or other operations, the underlying soils would be highly vulnerable to erosion by water. The plains support only a few hardy plants (e.g. Bindyi, *Sclerolaena* spp.) (Figure 2-11), except after rain.

Following rainfall, the gibber-gilgai landforms shed water to low-lying areas dissected by anastomosing channels (Wakelin-King 2011, 2014). At a micro-scale (10^1 to 10^2 m), the runoff flows to microrelief areas or gilgais with high water-holding capacity, shrink-swell, clay soils (2:1 layer silicates) supporting shrublands and grasses (salt-bushes, *Atriplex* spp.; Mitchell grasses, *Astrebla* spp.); these are refuges for animals and drought fodder for grazing stock. At a meso-scale (10^3 to 10^4 m), runoff from the gibber country determines where and how often the rivers flow, and drives their productivity.

The catchments generally conform to models of central Australian fluvial geomorphology, driven by occasional episodic flows and historic 'mega-floods' that reset geomorphic patterns at all spatial scales (Pickup 1991). In the study catchments, mega-flows along the Finke, Macumba and Neales have interacted to produce subtle differences in bedload composition, with implications for aquatic ecosystems (Nanson 2010; Wakelin-King 2011, 2014). For example (Wakelin-King 2014):

> *Surprisingly, though part of the same catchment, Arckaringa Creek's fluvial style is significantly different from that of the Neales River. Differences in geology and landscape processes mean that Arckaringa Creek is virtually devoid of refuge waterholes.*

As Figure 2-10 shows, still-active and older floodplains of the northern rivers intersect sandplain and dunefield landforms derived in part from Quaternary sands of the Simpson and Perdirka deserts. The rivers transport coarse sands and gravels, deposited as sandbars, small sandy rises or large-scale ripples (Pickup 1991; Wakelin-King 2014), depending on the energy of flows and the valley planform, which may be tens of metres to kilometres wide. Erosion and deposition of bedload sediments (1-5 m) prevent formation of deep channels necessary for the scouring flows that maintain waterholes (Boys and Thoms 2006):

> *The Finke River is known to have many waterholes which appear and disappear over time (Duguid, pers. comm. 2013) … small floods will tend to bring in sand, and large floods are needed to create the turbulence that scours them deep (Wakelin-King, 2014)*

There may be no permanent (Ark refuge) waterholes along the Finke, but a waterhole with a clay bed in the floodout supports water-dependent species (e.g. River Red Gum, *Eucalyptus camaldulensis*; Lignum, *Duma florulenta*), showing that water does persist for significant periods (Wakelin-King 2014).

The floodouts of the mid-lower Finke and lower Macumba rivers are termini for bedload transport, where the rivers enter a less-restricted valley planform (Pickup 1991). Their geomorphic setting implies that they receive infrequent, large river flows (Wakelin-King 2014) that would have undergone large transmission losses, recharging alluvial aquifers and deep soil-moisture stores that sustain vegetation in dry periods (Duguid 2011). These high-energy flows would also be diverted westward to wetlands like Snake and Allapalilla creeks (locations 's' and 'a' in Figure 2-10). The overflow wetlands are little-known, but support a rich diversity of plants and animals and may be refuges for terrestrial species (Eldridge and Reid 1998).

No permanent waterholes are known in the Macumba catchment, although fish monitoring suggests that a permanent (possibly bore-fed) refuge does exist (Schmarr et al. 2014). The Lindsay River in the upper Macumba catchment includes the Eringa chain of waterholes, lined by River Red Gums (Figure 2-4; Site 6, Figure 2-10). The waterholes occupy single channels with a low width: depth ratio and a stony, erosion-resistant bed, and are surrounded by highly-dissected gibber slopes in a narrow valley planform (Table 2-3).

Unlike the northern rivers, the Neales and Peake rivers have narrow valley planforms in gibber-gilgai country (Table 2-3; Figure 2-10) and flow over a fine-textured, stony, erosion-resistant bedload. Their floodplains more regularly receive high-energy, turbulent flows to scour and maintain waterholes (Wakelin-King 2011). Tree-lined waterholes are well-developed in converging reaches of the upper Neales and downstream of the Peake-Dennison Ranges (Wakelin-King 2011). In the mid- to lower reaches, broader valley planforms with alternating single-channel reaches (Algebuckina, Peake Gap) are associated with small floodplains. Along the mid-Neales and Peake, these reaches are associated with broad clay flats and spring complexes (Nilpinna, Freeling, Mt Dutton). Although the springs are above the floodplain level, they contribute diffuse saline discharge that affects soils and downstream refuges (Costelloe et al. 2005b).

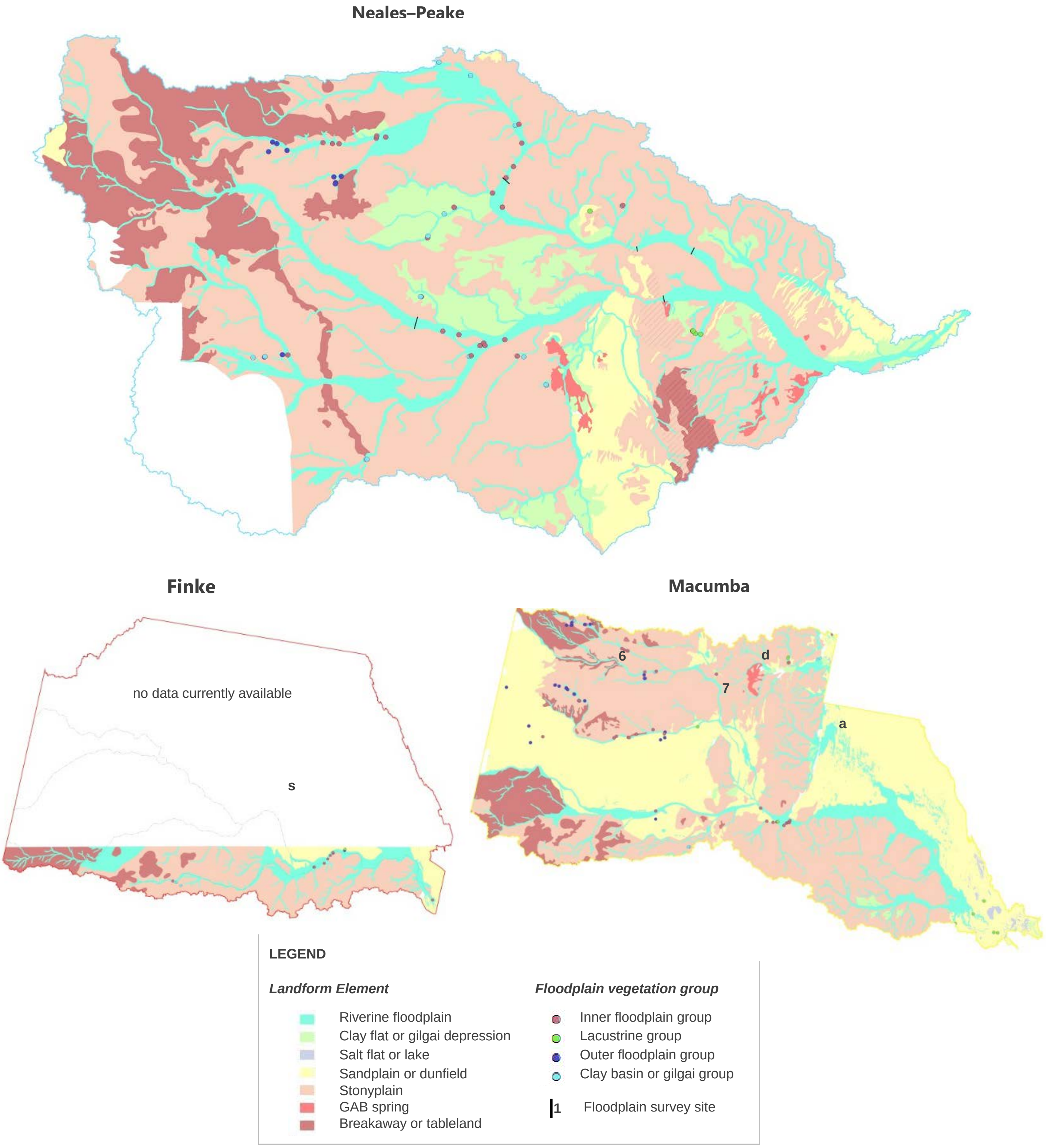
The Arckaringa Creek reach of the Neales-Peake catchment is a network of channels in a broad floodplain (Table 2-3) over deeply-weathered, readily transportable materials that prevent formation of persistent waterholes. Although both Arckaringa and Lora creeks (tributary to the Peake; Site 1, Figure 2-10) show waterhole-like features (for example, splays exiting convergent channels and deeper channels supporting riparian vegetation), surveys have not located significant aquatic habitats. The fluvial mechanisms are like those in the northern rivers (Wakelin-King 2014):

> *The small size and scarcity of 'waterholes' in Arckaringa and Lora Creeks is due to the constant bedload transport down the channels. Firstly, the presence of bedload during flow events dampens turbulence, reducing the likelihood of the macroturbulent scour which is responsible for waterhole formation. Few wide, deep channel segments will be created; where they do occur they will not be as big as those of the Neales River. Secondly, bedload fills up the channels, so any waterhole-shaped channel segment is unable to maintain space for free water.*

Table 2-2: Proportions of major landforms in WRLEB catchments

Catchment	Landform element proportions (Figure 4.8)							Area (10^3 km^2)
	Riverine floodplain	*Clay flat or gilgai*	*Salt flat or lake*	*Sandplain or dunefield*	*Stony plain*	*GAB spring*	*Breakaway or tableland*	
Finke	0.23	<0.01	<0.01	0.11	0.53	<0.01	0.13	3.6
Macumba	0.12	0.01	<0.01	0.37	0.42	<0.01	0.07	37.7
Neales	0.15	0.08	<0.01	0.07	0.57	0.01	0.13	27.6

Figure 2-10: Major landforms in WRLEB catchments (BDBSA 2014)[3]



[3] *Water features are a=Macumba overflow wetland (Ambullinna Waterhole), d=Dalhousie Springs complex, f=Freeling Springs complex, m=Mt Dutton Springs complex, n=Neales overflow wetlands (Nappamurra lakes), o=Nilpinna Springs complex, s=Finke overflow wetlands (Snake Creek)*

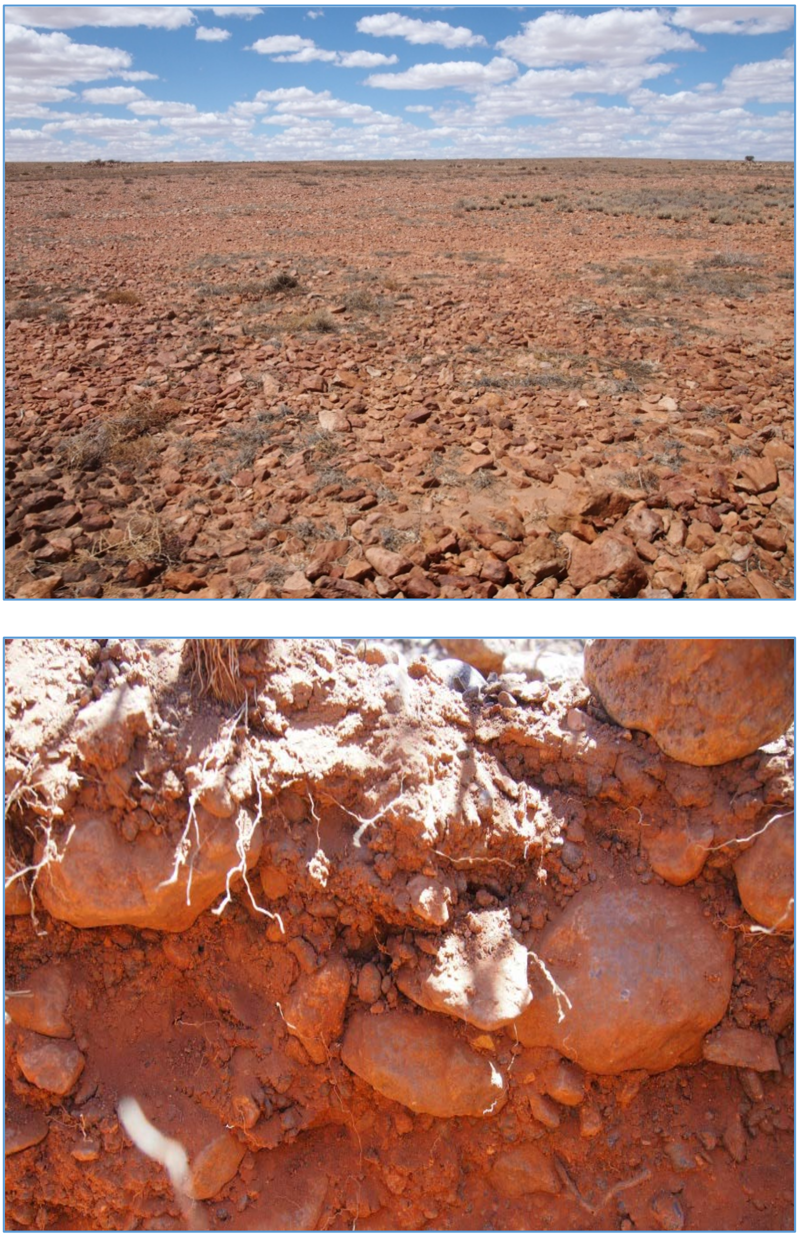


Figure 2-11: Gibber-gilgai landforms in the WRLEB

Table 2-3: Characteristics of the Neales–Peake and Macumba catchments

Catchment	Site	No.*	East	North	Elev.	Width (appr.) [m]	Depth (meas.) [m]	Slope (appr.) [deg]	Channel type	Refuge type
Macumba	Ethawarra		464297	7038536	211			0.07	single	Disco
	Eringa	6a	472475	7092759	228	50, 140	5, 1	0.16	single, overflow	Disco
	Carpamoongana	6b	475965	7041071	196	52, 100	6.5, 2.5	0.08	single, overflow	Disco
	Stevenson R	7	529035	7064387	151	550	0.5 - 2		multi-thread	None
Neales	Cramps Camp		537757	6935974	98	911	2 - 4	0.01	anastomose	Disco
	Stewarts	2	537845	6937648	98	650	2 - 4	0.01	anastomose	Disco
	Shepherds		542040	6951905	112			0.02	anastomose	Disco
	Hookeys		542932	6947455	113			0.07	anastomose	Disco
	Algebuckina	3	581306	6914207	63	48, 840	7, 1.1	0.03	single, overflow	Ark
	South Cliff	4	598062	6912581	50	1700	2 - 4	0.04	anastomose	Disco
	Tard		612019	6900806	39			0.03	anastomose	Extinct
	Retard		613347	6899978	40			0.07	anastomose	Polo
Peake	Arckaringa Ck	1a	466006	6924614	162	900	0.5 - 2	0.07	multi-thread	None
	Arckaringa Ck	1b	508131	6889467	101	2400	0.5 - 2	0.04	multi-thread	None
	Cootanoorina		530312	6883551	80			0.02	multi-thread	Extinct
	Mid Peake R		555142	6890942	67			0.04	no defined channel	None
	Mid Peake R		555165	6890981	66			0.04	no defined channel	None
	Peake Crossing	5	578772	6898695	51	50, 828	3, 1.5	0.00	single, overflow	Polo
	Warrawaroona/ Baltacoodna		588995	6897651	57			0.01	single	Polo

*Numbered sites in Figure 2-10

3. Indicators and response types for impact assessment

3.1 Introduction

The potential effects of flow regulation in the LEB have been emphasized in relation to water-resource development (Walker et al. 1997; Kingsford et al. 2014). While catchment-scale floods drive booms in productivity, the smaller 'bridging' flows that connect reaches and habitats or sustain productive patches through dry periods are no less important (Miles and Risby 2011; Costelloe and Russell 2014). The effects of regulation therefore may cover a wide range of spatial and temporal scales.

Assessments of mining impacts require an understanding of the significance of natural flow variability for species attuned to the 'boom and bust' dynamics of the region, including terrestrial as well as aquatic species (e.g. Brandle 1998; Eldridge and Reid 1998; Brandle et al. 1999; Gillen and Reid 2013). To identify indicators and their likely responses to changed regimes, it is necessary to identify the 'drivers' that structure communities and habitats in space and time. This is a basic question of ecology (Krebs et al. 1994): 'what drives the abundance and distribution of species?'.

In this section, 'multiple lines of evidence' are drawn from survey and monitoring programs in the WRLEB during 1999-2013. As mentioned, vegetation and fish are the only biotic groups for which there are sufficient spatial and temporal data for analysis, alongside the available hydrological data.[4] This information is used to test and refine the conceptual models described earlier (Section 1.4), leading to identification of potential indicators and response types that could be used to assess the impacts of CSG and mining development (Section 5.1).

3.2 Focus reaches

The study reaches are in areas under the direct or indirect paths of mining, from tenement data for the Arckaringa subregion (Department of State Development, unpublished). Four reaches were selected in the Neales–Peake catchment, namely Arckaringa Creek, Mathieson–Neales Corner, Algebuckina–Tardekarinna and Peake–Confluence (Figure 3-1). These were chosen on the basis of available hydrological and biological data (Costelloe et al. 2005a,b; Costelloe 2011; Cockayne et al. 2012; Montazeri and Osti 2014), and encompass the major geomorphic settings (cf. Boys and Thoms 2006). Other data to characterise hydrology and habitats were from cross-sectional surveys (May 2013, February 2014) of floodplains and landform-vegetation boundaries and waterbodies (e.g. farm dams, bore drains) that are not routinely monitored (Appendices A–B). Finally, additional data from eastern LEB catchments were included for comparison.

Refuge habitats along the upper Neales River, represented by the Mathieson-Neales Corner reach, include ephemeral to semi-permanent waterholes in poorly-channelized areas. Redirection of flows through the deeper (2-4 m) confined waterholes (Costelloe 2011) results in greater discharge, and 'mini-floodouts' (Wakelin-King 2011) occur immediately downstream of refuges. In contrast, the upper Peake River, represented by Arckaringa Creek, is akin to the bedload streams of the northern rivers. Frequent sediment redistribution there prevents the formation of deep waterholes (Wakelin-King 2014), but the shallow, anastomosing channels support a productive 1-3 km wide floodplain that is highly valued by pastoralists. The mid-reaches of the Neales and Peake rivers receive diffuse discharge from springs, increasing the salinity of downstream reaches and waterholes (Costelloe et al. 2005b). The Peake River receives inputs from the Nilpinna Creek and springs complex, and the Neales River is affected by the Ockenden Creek springs (Mt Dutton). Some poorly-channelized areas and floodplains in these reaches support low chenopod shrublands and salt-tolerant grasslands.

Subsequently, the Neales passes through the confines of the Peake-Dennison Ranges and enters gorge-like rock channels supplying two refuge waterholes, namely Algebuckina on the Neales and Peake to Warrawaroona on the Peake. Algebuckina is deeper (about 4 m cease-to-flow depth, 7.9 m bankfull: Costelloe 2011) and fresh to slightly saline; it is the only permanent Ark

[4] Other biota and abiotic processes may be more sensitive to changes arising from CSG and large coal mining impacts than fish and flora, however there was insufficient data for the study region to undertake any analysis for the purposes of this report.

refuge (Section 2.3) in the western rivers. In contrast, Peake to Warrawaroona is a chain of shallow waterholes (1-3 m cease-to-flow depth, 4-6 m bankfull: Costelloe 2011) that contract to hypersaline pools during drought.

Outflows from these refuges support 1-2 km wide floodplains down to the Neales–Peake confluence. Floodplains in the Algebuckina reach support salt-tolerant shrubs and grasses and woodlands, and those of the Peake support salt-tolerant grasslands including areas under pressure from grazing and erosion (Wakelin-King 2011). Saline and hypersaline reaches occur downstream of the confluence, and the river is poorly channelized or meanders through incised channels with little plant cover. The lower reaches extend from Algebuckina to Tardekarinna Waterhole, and from the Peake Crossing to the confluence.

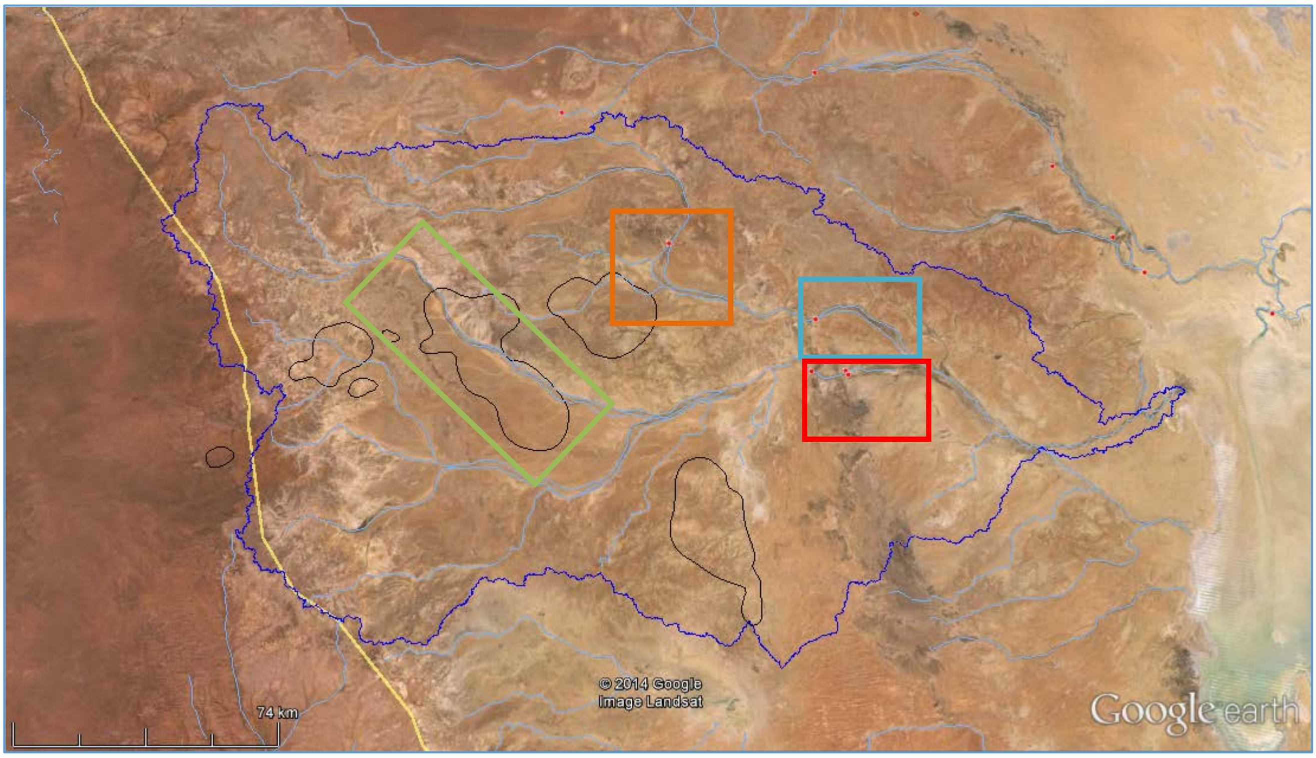


Figure 3-1: Study reaches in the Neales–Peake catchment

(green = Arckaringa, orange = Mathieson–Neales Cnr, blue = Algebuckina-Tardekarinna, red = Peake-Confluence), refuge waterholes (red dots) and coal deposits (black lines) (Department of State Development, unpublished)

3.3 Data and methods

3.3.1 Sources

Gridded rainfall data were obtained from the CSIRO *Australian Water Availability Project* (AWAP)[5]. Hydrological and ecological data were from various (mostly South Australian) natural resource databases, incorporating biological inventory surveys (e.g. ARIDFLO, Biological Databases of South Australia, SAAL Critical Refugia), the Lake Eyre Basin Rivers Assessment and other monitoring studies (e.g. Lake Eyre Basin Rivers Monitoring). Data for hydroecological modelling were from a rainfall-runoff model for the Neales–Peake catchment (Montazeri and Osti 2014). More details are in Appendix A.

Fish, vegetation, patch quality (Table 3-3) and water quality data were collected as described in Appendix B. Where water quality was measured at different depths, an average was taken. All categorical variables were assumed to have been accurately recorded, but these have been made over a number of years and there may be differences between observers.

[5] http://www.bom.gov.au/jsp/awap/

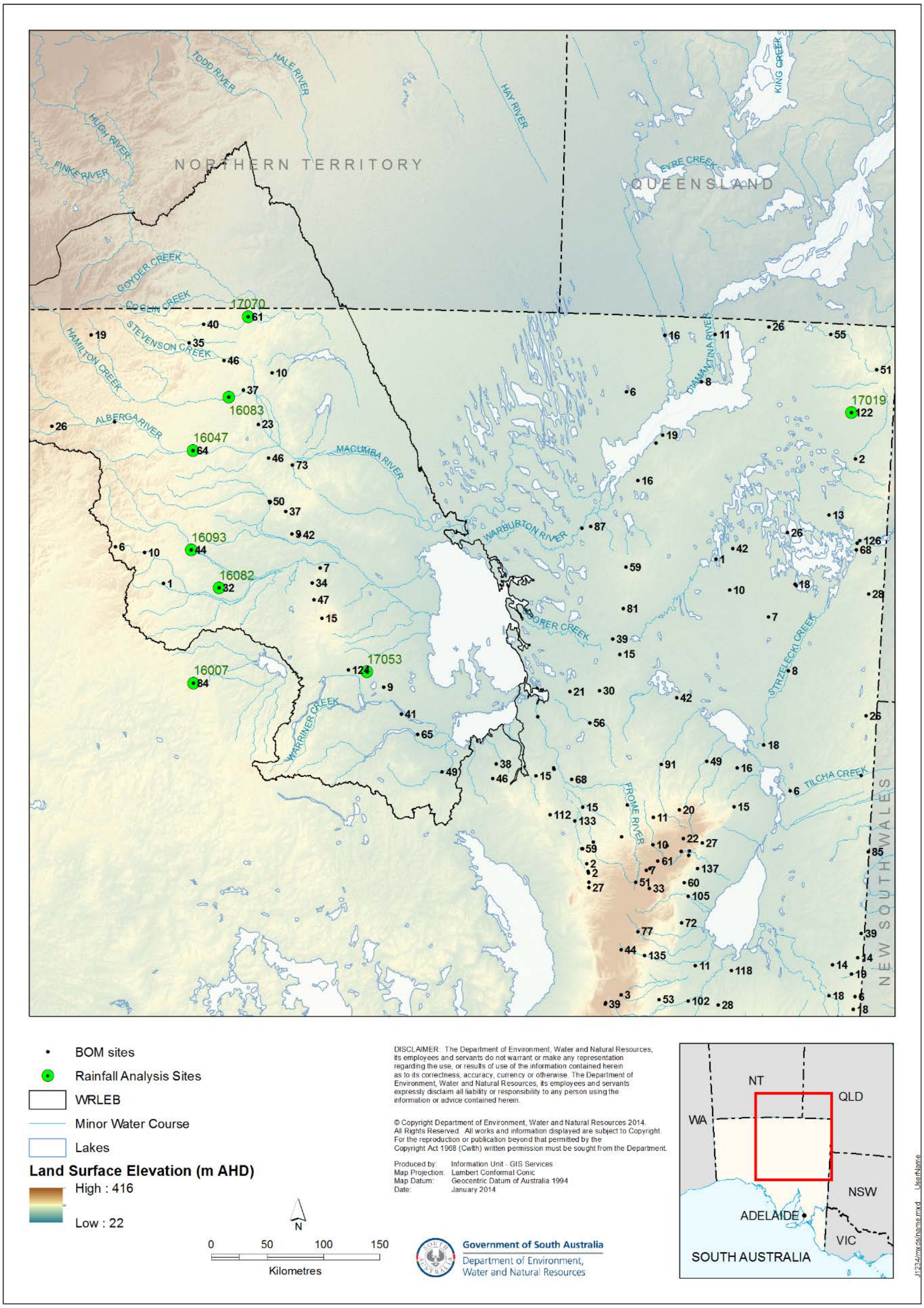


Figure 3-2: Locations of BOM/AWAP stations (numbers = years of recorded rainfall)

3.3.2 Hydrological analysis

To investigate changes in rainfall, residual mass curves were constructed for eight AWAP sites (Figure 3-2). Monthly residuals were calculated for 1950–2013, standardised by mean annual rainfall and scaled by a factor of 1000. Time series of cumulative monthly residuals for each location were plotted to represent spatial and temporal patterns predicted by the AWAP model.

To investigate indicators and response types, event-based analyses (Knighton and Nanson 2001) were undertaken at catchment, reach and patch scales, using recent records and modelling data (Montazeri and Osti 2014). Methods included satellite image and time-series analysis and stage-discharge curve modelling. Axes of flow variability were also characterized.

3.3.2.1 Catchment-scale hydrology

To characterize catchment-scale hydrology, remote-sensing derived vegetation and water indices were combined for the Neales–Peake, Macumba and lower Finke catchments. Free water and vegetation greenness indices were combined as dual indicators of flow and flood extents, the latter capturing the green flush that remains following major flow events. The use of vegetation indices as a quantitative indicator of flow events has been trialled to offset biases in using free-water indices to map episodic flows in channels that are obscured by vegetation (Stewardson et al. 2009; Zhuang et al. 2011). By combining free water and vegetation greenness, variations in flow and flooding could be captured while ensuring that small or localized flows were not missed between satellite orbits (26–30 d).

Eight dates were analysed, corresponding to flow and no-flow periods of varying seasonality and intensity (Table 3-1). Imagery was sourced[6] for five scenes encompassing the WRLEB (Appendix A, Figure 6-2). Image analysis used the Normalized Difference Vegetation Index (NDVI) (B4–B3/B4+B3) and Normalized Difference Water Index (NDWI) (B2–B5/B2+B5). NDVI indices were adjusted to delineate known open-water bodies (Figure 3-5), and indices were thresholded at > 1 to correct for seasonal differences between scenes, using restricted range standardization (Table 3-1). Positive NDVI values indicate photo-synthetically (PHS) active vegetation. As seasonal differences in PHS occur independently of moisture, *a priori* restricted ranges were applied to a standard transform (range 0–220) to normalize NDVI ranges according to the season of capture. For example, during drought a lower minimum NDVI score of 0.02 was used to capture low PHS activity, expected for perennial vegetation that maintained low transpiration, drawing on residual moisture (i.e. deep soil moisture or alluvial groundwater). Sample results and validation windows are shown in Figures 3-3 and 3-4 for no-flow and flow conditions, respectively.

Table 3-1: Landsat TM scenes and relative magnitudes of captured no-flow and flow events

Flow Stage*	**Date DD/MM/YYYY** (last in scene capture)	**Modelled flow conditions†**			**Restricted range ‡ (min-max NDVI score)**
		Dry spell (zqd.spell)	*Flow duration (qd.90)*	*Flow volume (qv.90)*	
No Flow (I)	9/03/1999	67	0	0	0.05 – 0.5
Catchment (II-C)	6/05/2000	0	85	198	0.1 – 0.5
Drought (IV)	17/01/2001	231	0	0	0.02 – 0.3
sub-catchment (II-S)	14/09/2001	30	60	244	0.02 – 0.3
sub-catchment (II-S)	28/03/2003	0	34	196	0.02 – 0.3
Drought (IV)	10/03/2008	391	0	0	0.02 – 0.3
Local (II_L)	26/05/2010	0	55	7	0.1 – 0.5
Catchment (II_C)	27/04/2011	0	80	461	0.15 – 0.5

*Defined in the ARIDFLO model (Section 1.4.6, Figure 1-3)

†Modelled flows after Montazeri and Osti (2014); zqd.spell=length of dry spell in days; q90=flow volume (GL) and duration (d) in 90 days prior to last date of scene capture (standard for metrics calculated at Algebuckina waterhole).

‡NDVI range standardized to 0–220.

[6] http://earthexplorer.usgs.gov/

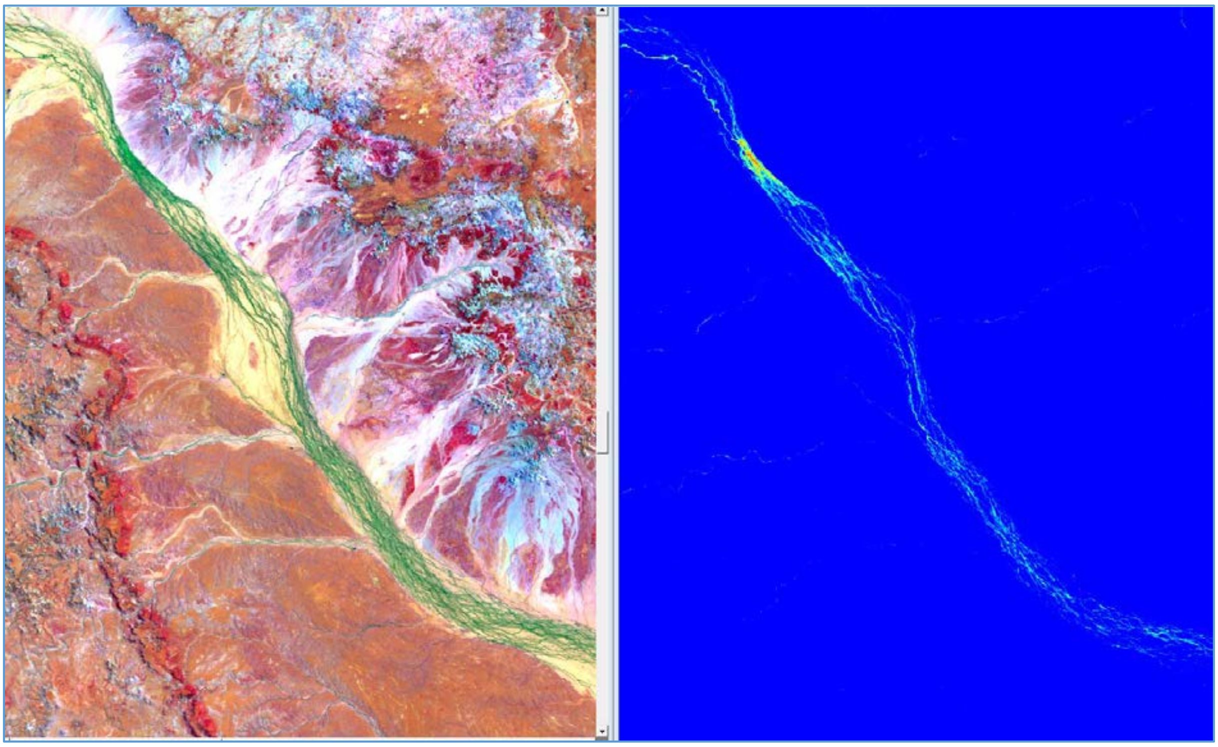


Figure 3-3 (top): Sample NDVI classified image[7] using restricted range standardization (Arckaringa Creek)

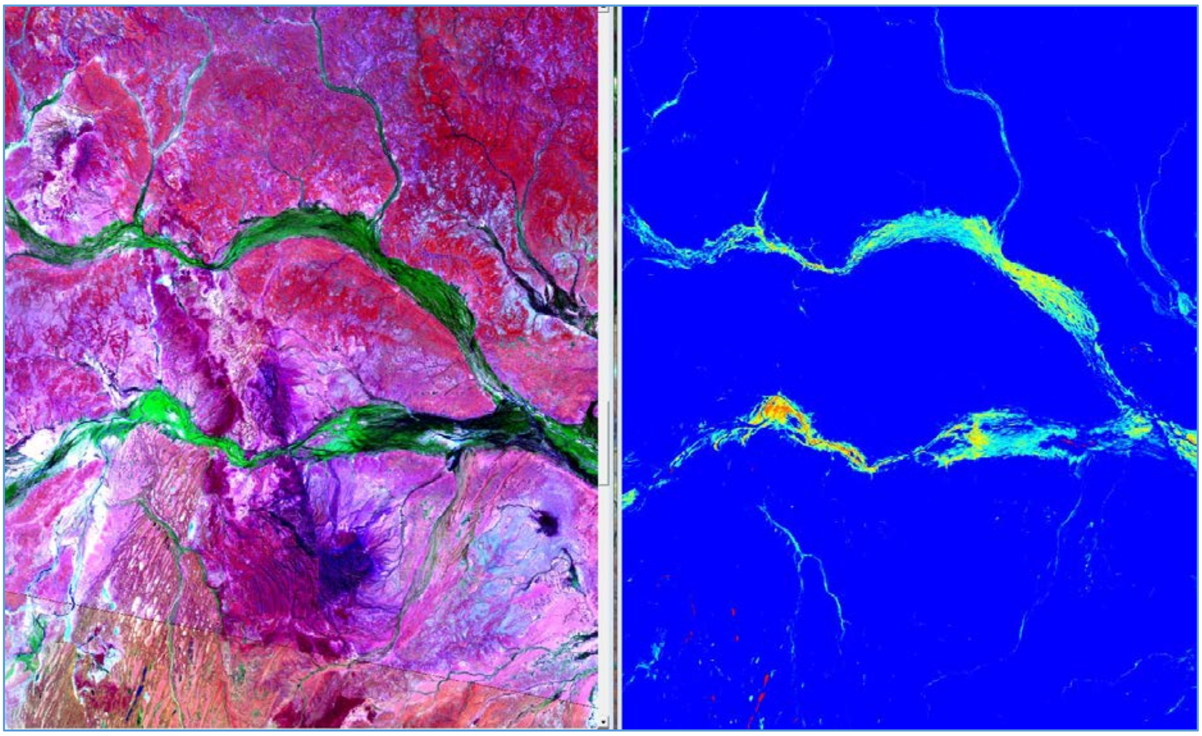


Figure 3-4 (bottom): Sample NDVI classified image[8] using restricted range standardization (Neales catchment, Peake Confluence)

[7] *The image shows Arckaringa floodplain in March 2008, one of four windows per scene used to validate the image classification. The left panel is a 7-4-1 false colour image; the right panel is the NDVI classified image.*

[8] *The image shows the Neales-Peake Confluence floodplain in May 2011, one of four windows per scene used to validate the image classification. The left panel is a 7-4-1 false colour image; the right panel is the NDVI classified image.*

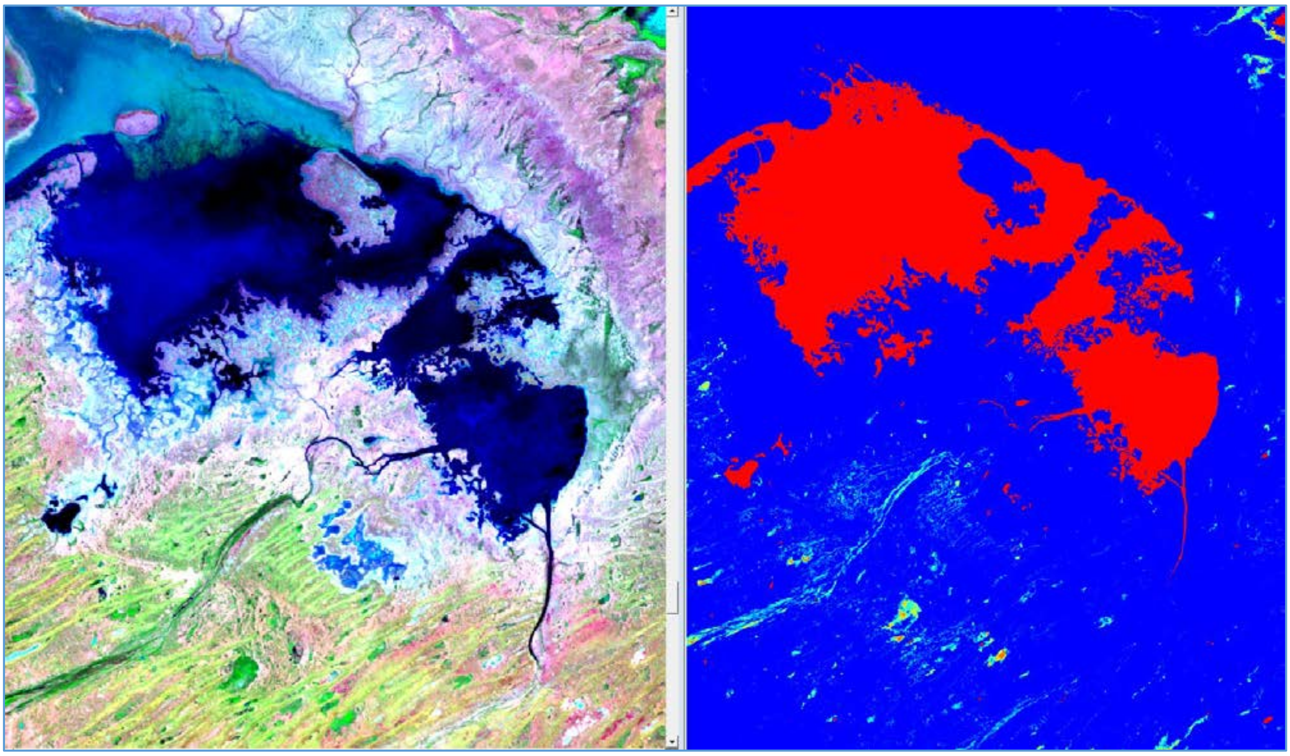


Figure 3-5: Sample NDWI threshold image classification[9] (Lake Cadibarrawirracanna)

3.3.2.2 Reach-scale hydrology

Flood regimes for each reach were estimated from the areas of inundated floodplain. Satellite imagery representing no-flow and flow periods was processed for each date (Section 3.3.2). From the images, relationships were developed between modelled metrics (Montazeri and Ozti 2014) at the outlet of each reach and downstream areas of floodplain inundated. Inundation extents for each scene/date were calculated as counts of vertices intersecting active flooding (NDWI >0) or greenness (NDVI >0.02) divided by the total vertices intersecting the floodplain on a regular grid. Areas of maximum potential floodplain were demarcated as areas between upstream and downstream locations connected by high flows (Figure 3-6). These relationships are appropriate given the uncertainty in modelled flow estimates (Montazeri and Osti 2014).

Modelled time-series flow and water-level data were further investigated for the refuge waterholes and floodplains in the focal reaches of the Neales–Peake catchment (Figure 3-1). Mean flows (±SE) were compared for each location at annual and seasonal scales. Modelled discharge (ML/d) and water level (m) data were plotted as time series from 1999–2013 to identify the incidence and duration of local and sub-catchment flows and amplitude (water level) fluctuations predicted by the Neales–Peake rainfall–runoff model (Montazeri and Osti 2014). In addition, modelling data were used to estimate the frequency and duration of connecting flows between upper and lower reaches, or sub-catchment flows, using discharge: loss ratios, where connecting flows were implied if the net flow in a link exceeded 1 ML/d (the difference between modelled flow and losses). Transmission losses (L) were calculated according to Montazeri and Osti (2014). Discharge (D) was directly output at the upstream storage, with no attempt to calculate the direct contribution from rainfall on the connecting reach (this would be only a small proportion of connecting flows).

[9] *The image shows Lake Cadibarrawirracanna in May 2011, one of four windows per scene used to validate the image classification. The left panel is a 7-5-3 false colour image; the right panel is the NDWI image classification.*

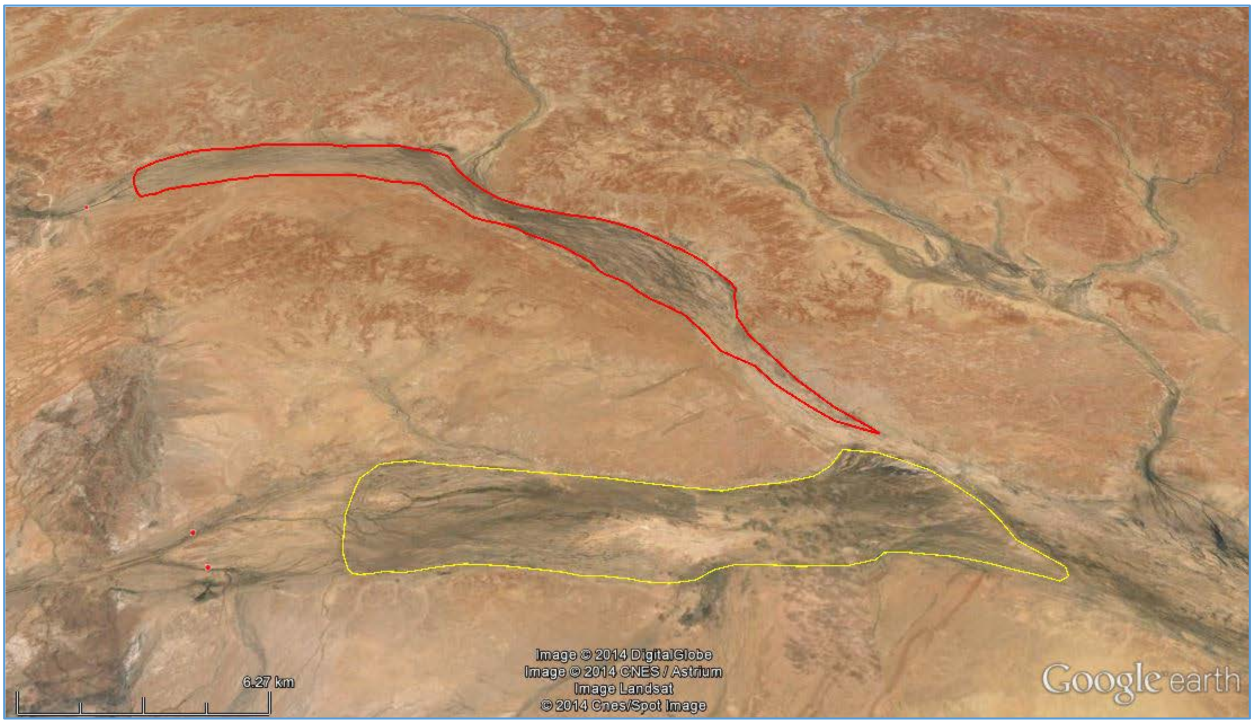


Figure 3-6: Floodplain areas (red= Algebuckina-Tardekarinna; yellow= Peake-Confluence)

3.3.2.3 Axes of flow variability

To investigate hydrologic variability, 29 flow metrics were calculated from modelled flow and water-level data for the Neales–Peake catchment (Table 3-2). The metrics were calculated for site-visit data coinciding with fish monitoring (Cockayne et al. 2012). The samples (2007–12) captured a full range of flow stages, in accord with the ARIDFLO model (Section 1.4.6); site visits captured an extended drought from 2007–09 (flow stage IV) followed by local flows in 2009-10 (stage II_L) and a catchment flood in 2010–11 (stage II_C), followed by cease-to-flow (stage III) and no-flow stages from 2012–13 (stage I). Principal Components Analysis (PCA) using the rda function in the *Vegan* package for *R* (Oksanen 2011) was used to partition variance among sites by visits, with axes defined at a cumulative variance threshold of 82.7% (the first three PC axes). The three axes were redefined in accord with conceptual models (Section 1.4), and indicator metrics were identified from the relative strengths of axes (explained variance) with covariance relationships (Olden and Poff 2003).

Table 3-2: Metrics used to classify axes of flow variability, with scaled Principal Component (PC) scores

Flow facet	Definition of metric	Units	Abbr.	PC1	PC2	PC3	PC4
Flow (duration)	Duration of present flow	d	days.q.spell	0.62	0.38	-0.38	0.19
	Days of flow in last 90 d	ʻʼ	qd.90	0.93	-0.27	-0.38	-0.18
	Days of flow in last 180 d	ʻʼ	qd.180	0.94	-0.16	-0.43	-0.14
	Days of flow in last 365 d	ʻʼ	qd.365	1.13	0.15	-0.06	-0.06
	Maximum duration of flow in last year	ʻʼ	qd.max.yr	0.94	0.27	-0.16	0.26
	Duration of present dry spell	ʻʼ	zqd.spell	-0.86	0.00	-0.06	0.15
	Maximum duration of dry spell in last year	ʻʼ	zqd.max.yr	-0.74	0.50	0.59	0.03
Flow (volume)	Volume of present flow	ML	vol.q	0.10	-0.21	-0.39	0.28
	Volume of flow in last 90 d	ʻʼ	qv.90	0.94	0.18	0.23	-0.38
	Volume of flow in last 180 d	ʻʼ	qv.180	0.94	0.20	0.23	-0.32
	Volume of flow in last 365 d	ʻʼ	qv.365	0.98	0.28	0.44	-0.16
Pulse (event)	Average daily flow during last flow event	ML/d	qve	0.94	0.19	0.54	-0.20
	Duration of last flow event	d	qve.spell	0.77	0.47	0.19	0.30
	Deviation of last flow event (z score)	[-]	qve.st	0.96	0.23	0.49	-0.18
	Days of flow in a rising limb phase in last year	d	qs.rlm.365	1.06	-0.03	-0.20	-0.04
	Days of flow in a falling limb phase in last year	ʻʼ	qs.fal.365	1.11	0.22	0.00	-0.06
	Maximum duration of rising limb in last year	ʻʼ	qs.rlm.max.yr	0.68	0.29	-0.37	0.34
	Maximum duration of falling limb in last year	ʻʼ	qs.fal.max.yr	0.78	0.47	-0.19	0.28
Amplitude	Present water level	m	wl	0.43	-0.83	0.13	0.48
	Deviation of wl from cease to flow wl	m	wl.st	0.70	-0.50	-0.35	0.34
	Duration of present wl low (wl <20 % lower than ctf)	d	zwld.low.spell	-0.54	0.25	0.07	-0.63
	Duration of present wl high (wl >20 % higher than ctf)	ʻʼ	wld.high.spell	0.00	0.00	-0.01	0.01
	Days of wl low in last 365 d	ʻʼ	zwld.low.365	-0.52	0.00	-0.25	-0.69
	Days of wl high in last 365 d	ʻʼ	wld.high.365	0.19	0.10	0.06	0.00
Frequency	Flow events in last 90 d	count	qve.90	0.30	-0.80	-0.42	-0.33
	Flow events in last year	ʻʼ	qve.365	0.59	-0.58	0.06	-0.57
	Flow events in last 2 years	ʻʼ	qve.730	0.57	-0.64	0.20	-0.42
Size	Current waterbody volume	GL	wh.v	0.24	-0.75	0.65	0.39

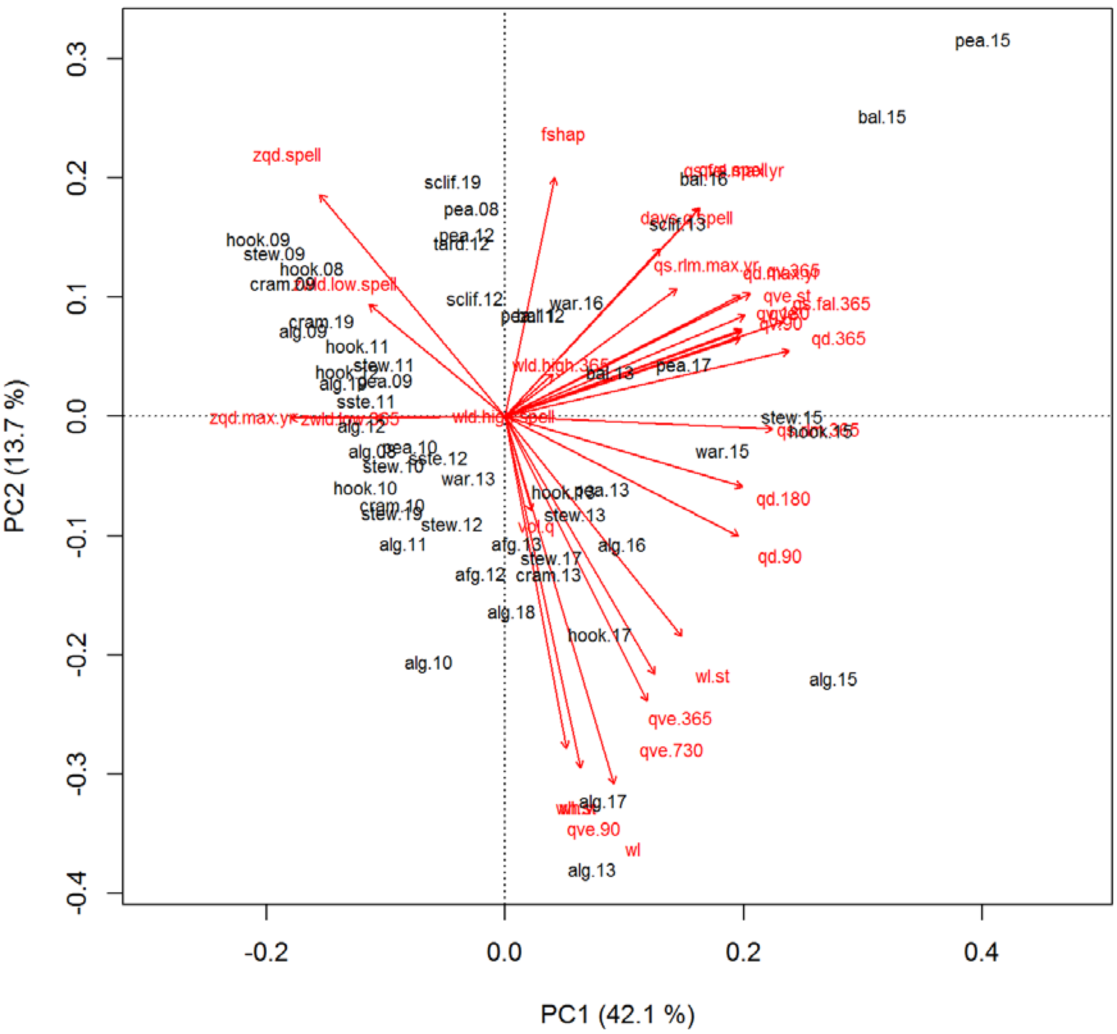


Figure 3-7: Principal Components Analysis of flow variability in the Neales–Peake catchment

Vector lengths are proportional to the percent variance explained across sites (site.visit) by each metric. For abbreviations, see Table 3.2.

3.3.3 Ecological analysis

Ecological analyses were undertaken to elucidate the spatial structure of aquatic and terrestrial assemblages from patch (among habitats) to landscape scales (among catchments), with regard for environmental variables. Non-metric Multi-Dimensional Scaling (NMDS) was combined with Generalized Additive Modelling of environmental surfaces ('ordisurf' and envfit models in the *Vegan* package for *R:* Oksanen 2011).

NMDS plots were constructed from presence-absence data for freshwater fish from 2007–12, recorded on 13 visits to 42 sites across the LEB (Appendix A). At the landscape scale, waterhole assemblages (*n* = 110) at 30 sites were analysed across the Cooper (coo, *n* = 19), Diamantina (dia, *n* = 9), Neales–Peake (nea, *n* = 58), Finke (fin, *n* = 12) and Macumba (mac, *n* = 12) catchments. At the patch-scale, 73 natural and man-made localities in the Neales–Peake catchment were analysed: waterholes (wh, *n* = 58), floodplains (in, *n* = 4), springs (sp, *n* = 5), bore drains (bo, *n* = 3), and dams (dm, *n* = 3). Waterhole assemblages (*n* = 58) at 38 sites in the Neales–Peake catchment were analysed separately for envfit modelling (see below).

NMDS plots were constructed for scaled abundances of perennial plants (modified Braun-Blanquet cover scale) in flood-prone environments. Due to a lack of prior flood-extent mapping, these environments (excluding springs) were defined by < 5 m elevation change and < 5000 m distance to nearest watercourse or wetland, based on Geographic Information System (GIS) processing (see below). Ephemeral species were excluded, to minimise the influence of different timing and conditions for biological survey data (Appendix A).

A landscape-scale assessment of riparian and flood-dependent vegetation by Brandle (1998) distinguished associations based on the relative dominance of species on floodplains, stony gilgais and sandplains (Figure 2-10). A re-analysis was undertaken, for perennial species only, to identify potential drivers for flood-dependent species. The base cut-off procedure using GIS was useful for identifying vegetation groups, indicator species and their environmental drivers including elevation, clay content and distance to watercourse (Appendix D.4). These landscape-scale patterns are inventoried elsewhere (Brandle 1998; McIlwee et al. 2013) and are not discussed further here.

To investigate the influence of flow and flood regimes on small-scale habitat zonation, sites on the floodplain and in flood-prone areas were analysed separately (Group 1, Appendix E.4). Relationships between assemblages and environmental variables were investigated using modelling functions in *R* (R Development Core Team 2011). Variables are listed in Table 3-3.

GIS variables (Table 3-3) were spatially derived in ESRI ArcGIS® (version 10.1) using *near*, *buffer* and *intersect* to identify geospatial determinants. *Extract* was also used to retrieve elevations at points along the river network using a 1-sec Digital Elevation Model. A spatial join was applied at each step to ensure that survey plot field tables were not amassed.

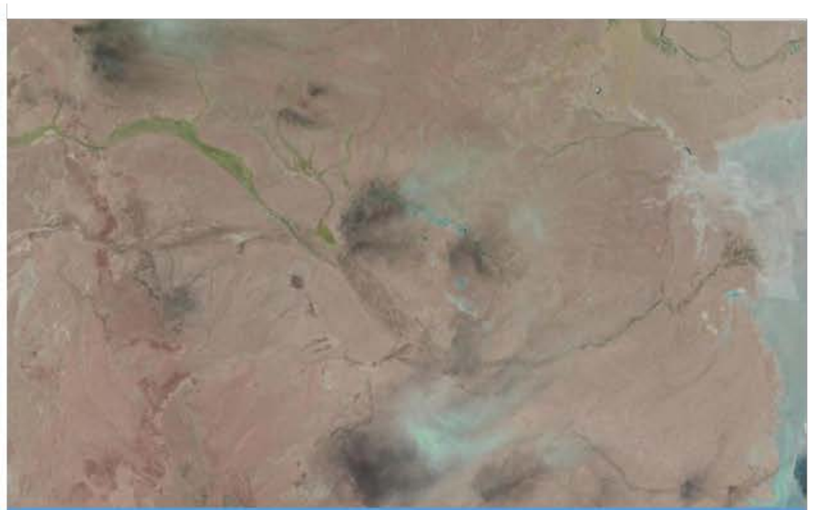
Table 3-3: Variables tested for relationships to aquatic and terrestrial assemblages

Aquatic	**Terrestrial**	
Water quality	***Patch quality***	***GIS variables***
Temperature (⁰C)	Site slope (º)	Elevation (m ASL)
pH	Strew size (mm)	Distance to nearest watercourse (m)
Salinity (ppt)	Strew cover (%)	Elevation of nearest watercourse (m ASL)
Dissolved oxygen (%)	Litter cover (%)	Distance to nearest open waterbody (m)
Turbidity (NTU)	Soil clay content (%)	Elevation of nearest open waterbody (m ASL)
Nitrate-N (mg/L)		Elevation change to nearest water (m)
		Distance to nearest waterhole (m)
		Landform element (1–6)

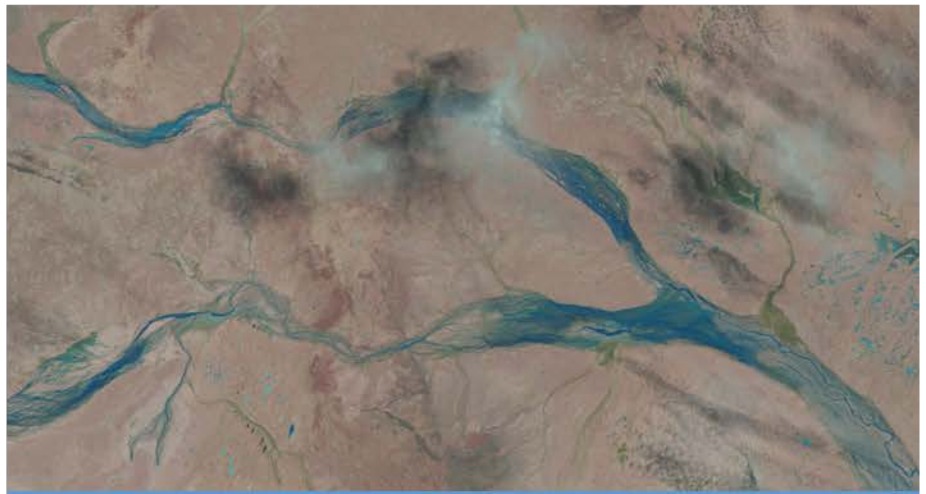
Environmental surfaces were interpreted in relation to indicator species and continuous variables using the envfit and ordisurf functions in the *Vegan* package for *R* (Oksanen 2011). Categorical variables (e.g. site, visit, wetland type, landform element) were analysed using permutational analysis of variance in the Adonis function of *Vegan*. Indicator species were determined using the indval function in the labdsv package (Roberts 2013).

Subtle shifts in the relative dominance of species on microrelief (a ‘non-standard floodplain’) show how hydrology interacts with other drivers in assembling floodplain communities. To investigate the flood regimes supporting non-standard flood-plains, a method was trialled for the four study reaches (Figure 3-14) using the average return interval (ARI) of stage heights (Montazeri and Osti 2014) to indicate inundation frequency. Preliminary results are in Appendix E.

Gathering summer storm over Neales river
(photo: J Vanlaarhoven)



Local rain showers, Neales river (Jan, 2003)
(source: http://landsatlook.usgs.gov/)



Catchment flood, Neales river (Feb, 2011)
(source: http://landsatlook.usgs.gov/)

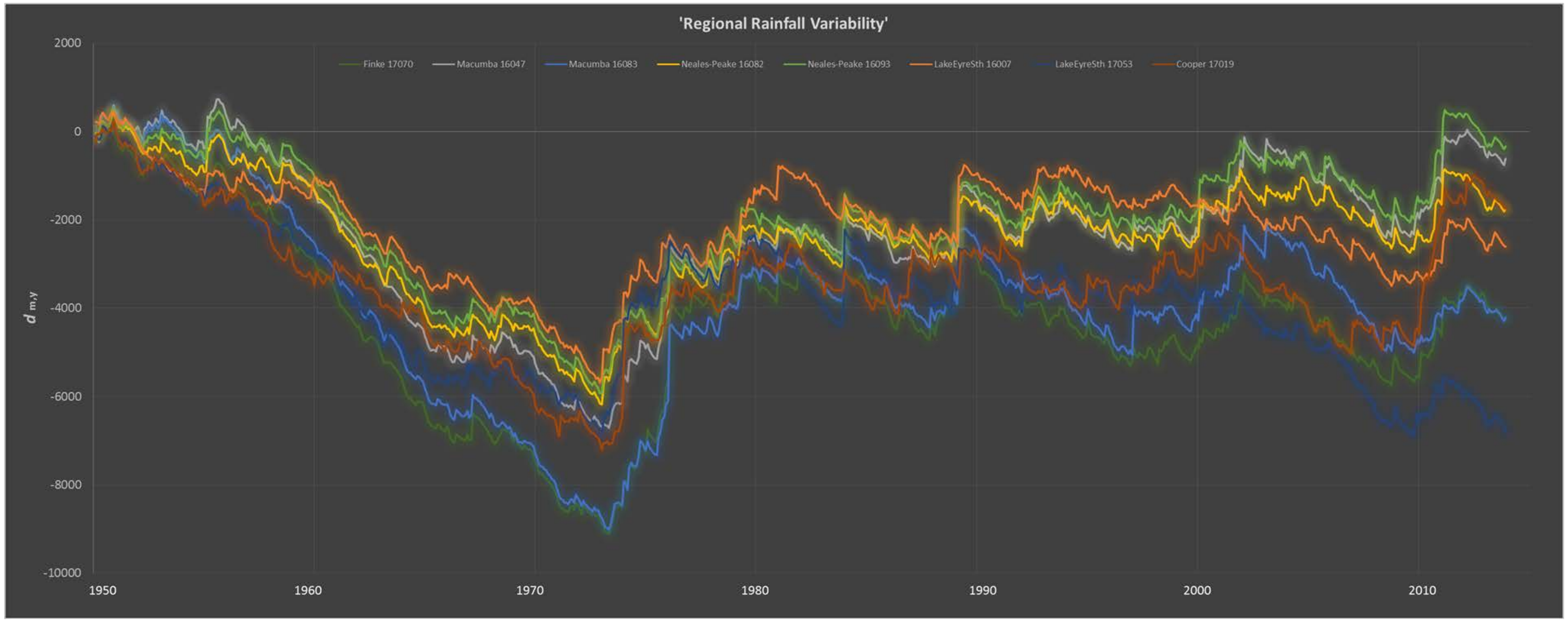


Figure 3-8: Time series of rainfall deviations (*d*) above average monthly rainfall, standardized for Mean Annual Rainfall (1000 units = MAR for locations in Figure 3-2)

3.4 Results and discussion

3.4.1 Rainfall

Time-series analysis of residual monthly rainfall shows local variability nested in a longer return interval (Figure 3-8). While there is variability between stations and years, major rainfall periods and droughts are more-or-less coincident between stations. Wet periods are shown by upward trends in residual monthly rainfall in the mid-1970s, late 1980s, early 2000s and from 2010–12, and dry periods are shown by downward trends from the mid-1950s to mid-1970s, 1980s to 1990s and mid-2000s. Wet periods coincide with filling of Kati Thanda–Lake Eyre (Kotwicki 1986), and major droughts are evident in monitoring from 2000–09 (McNeil and Schmarr 2009). Although localized thunderstorms and winter-rainfall events do occur (Costelloe 2011), larger rainfall pulses come from summer monsoonal influences linked to sea-surface temperatures (El Niño Southern Oscillation: Peel et al. 2002). The predictability of this relationship allows reconstruction of the palaeohydrology of the basin (Kotwicki and Isdale 1991).

Under a pulse-reserve model, more or less decadal regional rainfall drives productivity booms that contribute to energy reserves and redistribute water and nutrients, while short, localized rainfall supplies critical moisture to maintain patches through drought (Nano and Pavey 2013). Patch-interpatch dynamics provide refuges and continuity of resources for the biota, maintaining their capacity to respond to later floods. These spatial and temporal hierarchies shape the life histories of plants and animals in the region (Nano and Pavey 2013), and would influence the ways that ecosystems respond to flow-regime changes associated with mining or other disturbances (Stafford Smith and McAllister 2008; Arthington and Balcombe 2011).

3.4.2 Natural flow regimes

3.4.2.1 Western rivers floodplains

Satellite imagery for flow and no-flow periods in 1999–2011 (Figures 3-9 to 3-11) highlights the episodic nature of river flows in the WRLEB. Free water was rarely detected on the floodplain, although flow events did occur in this period. Short residence times (in relation to satellite return times), cloud cover, vegetative cover and the narrowness of open water areas (in relation to image pixel size) make Landsat imagery unreliable for detecting maximum flood extents (Stewardson et al. 2009). The more reliable signature for these rivers is activation of plant growth in response to flows, indicated by fluctuations in greenness (NDVI) between no-flow events and flow events of different magnitude.

Following catchment-scale floods in summer 2000 and 2011, free water and green-flushing were evident in all catchments. As mentioned, these responses follow flooding caused by approximately decadal rainfall pulses from subtropical low-pressure systems (Figure 3-8). The existence of free water in overflow wetlands (particularly Snake Creek) and across the landscape was obvious in the 2000 floods, when there were ‘booms’ in fish and waterbird populations throughout the basin (Costelloe et al. 2007). Free-water residence times were prolonged in these areas, and persisted even into the drought of 2001.

In contrast, the floods in 2011 brought a response from vegetation by April–May, with free-water remaining only in isolated locations that rarely receive flows (e.g. Macumba overflow wetlands, Simpson Desert inter-dunal wetlands). The response showed that the flows boosted productivity in almost all floodplain areas and connected reaches. The northern bedload rivers had much more extensive areas of vegetation (NDVI >0.3) compared to the Neales–Peake. Areas of high productivity were evident in the Finke floodout ‘forest’ and the Macumba floodout and overflow wetlands. On a smaller scale, patchiness in productivity was apparent on Neales River floodplains, where there were localized hotspots (NDVI >0.3) in the upper reaches of Arckaringa and Lora Creeks and the small floodouts downstream of Algebuckina and Peake Gap (yellow to orange hues: Figure 3-11). These spectacular events received international attention (Poulter et al. 2014).

Dry scenes (1999, 2001, 2008) indicate contracted, dormant floodplain communities and the productive areas (positive NDVI) corresponded more or less to the productive ‘pockets’ of floodplain seen under high-flow conditions. These areas are relatively isolated from refuge waterholes. Water from the Finke floodout, the Macumba overflow wetland (= Ambullinna Waterhole: Purdie 1984) and localized areas of the Arckaringa floodplain may sustain productivity in those areas, which include some refuges (Eldridge and Reid 1998). Groundwater inputs may be significant, but field studies are needed for confirmation.

Intermediate scenes (September 2001, 2003, 2010) showed some reaches receiving flows at locations and times when others did not. This is consistent with the hydrological modelling, showing that localized rainfall frequently results in sub-catchment- or reach-scale flows that do not connect localities throughout the catchment and are not well-represented by the sparse gauging network (Montazeri and Osti 2014; Ryu et al. 2014).

It is difficult to generalize a flow regime from only eight scenes, but some patterns are evident. Lower reaches in the Macumba and Finke receive more extensive flows than the Neales. The floodouts on northern rivers appear to support more frequent, extensive flows than previously recognized, and significant refuge habitats may exist in the Macumba overflow wetlands and floodouts. In contrast, the Neales received flows with no discernible pattern in its upper and lower reaches. Further work is required, but this interpretation is consistent with conceptual models linking sub-catchment and local flows to short periods of connectivity, supporting a shifting mosaic of productive patches across the landscape (cf. Stafford Smith and McAllister 2008).

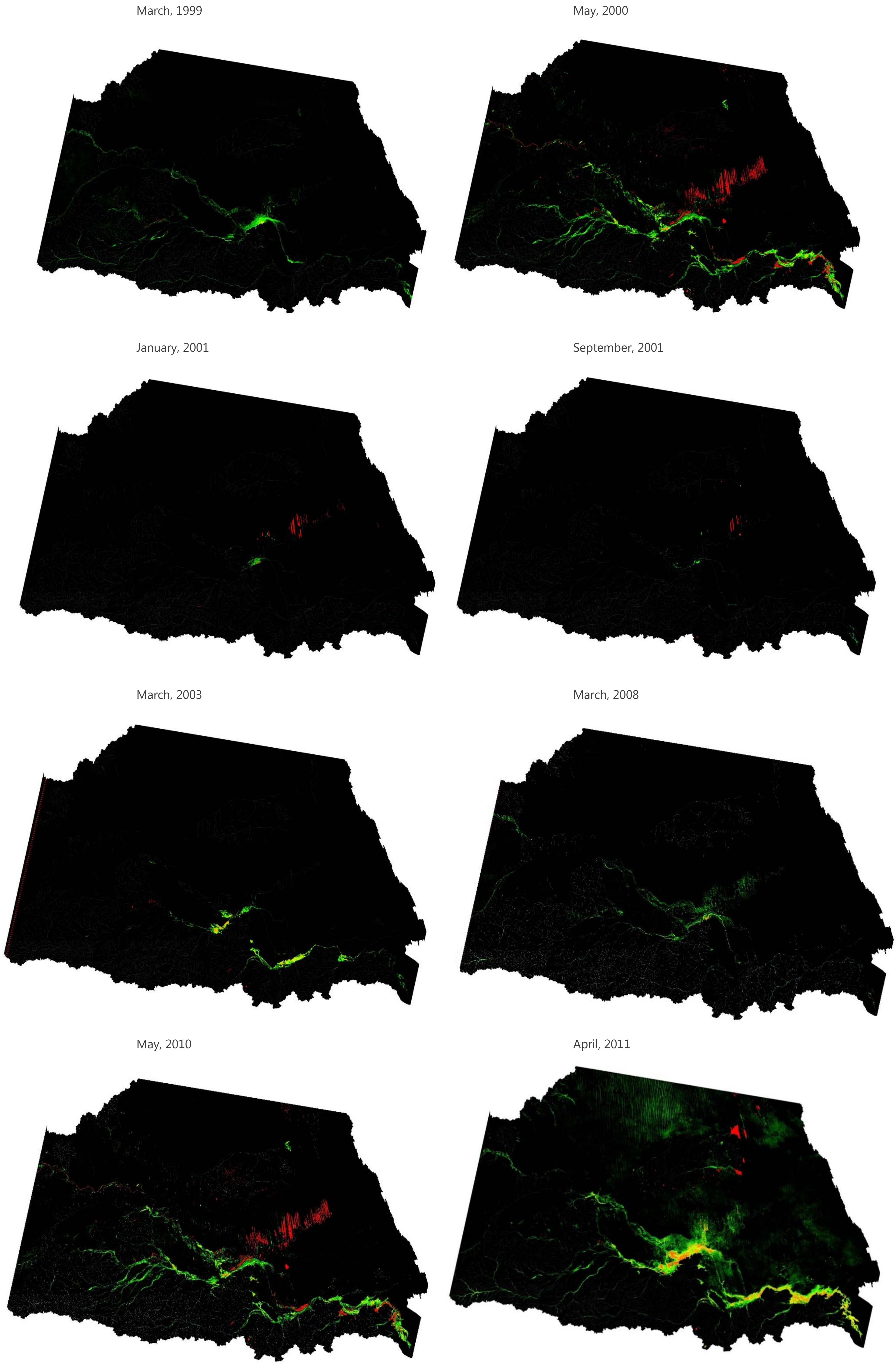


Figure 3-9: Catchment-scale hydrological variability in the lower Finke catchment (green to orange are NDVI 0.02–0.5; red is NDWI >0)

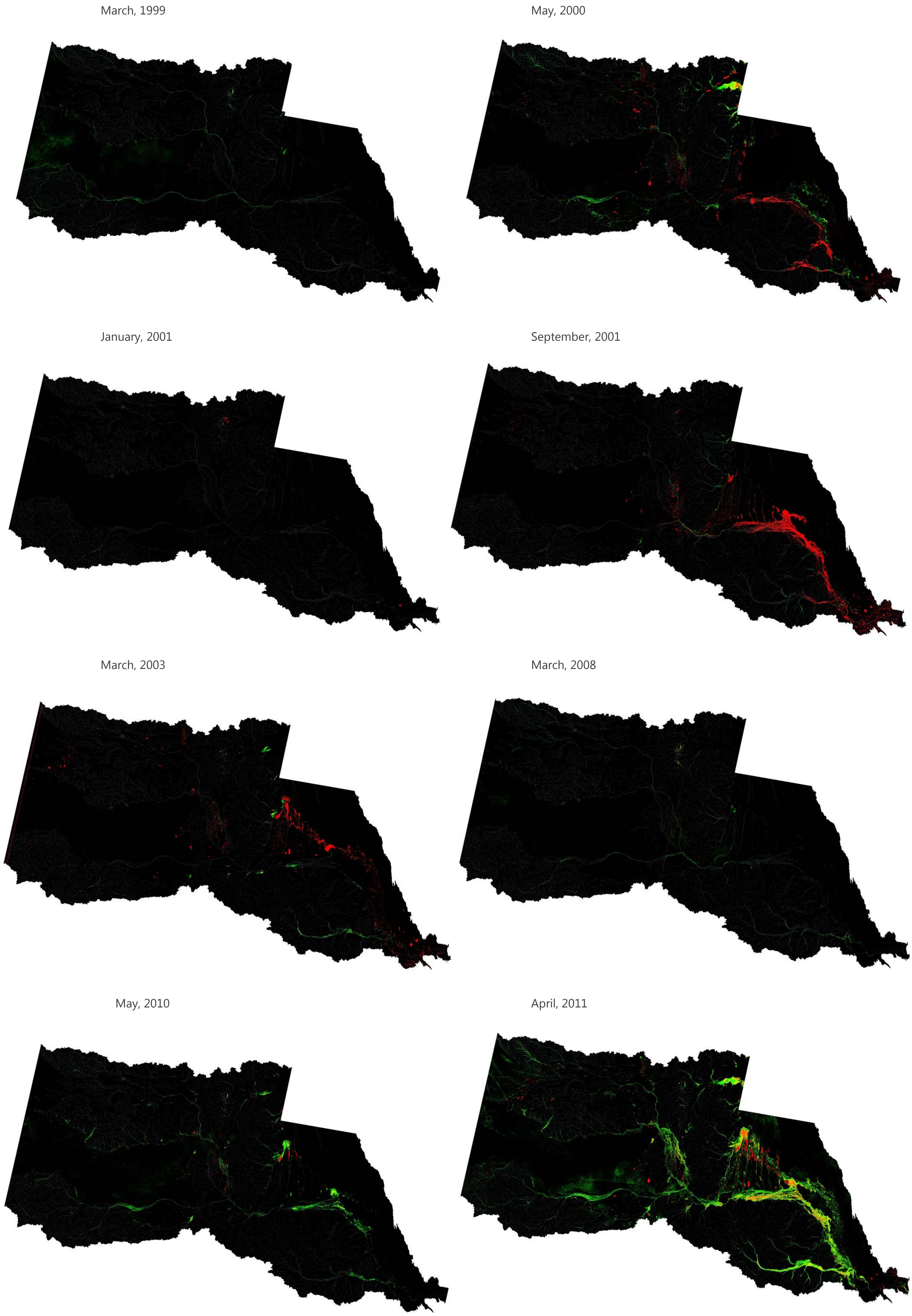


Figure 3-10: Catchment-scale hydrological variability in the lower Macumba catchment (green to orange are NDVI 0.02–0.5; red is NDVI >0)

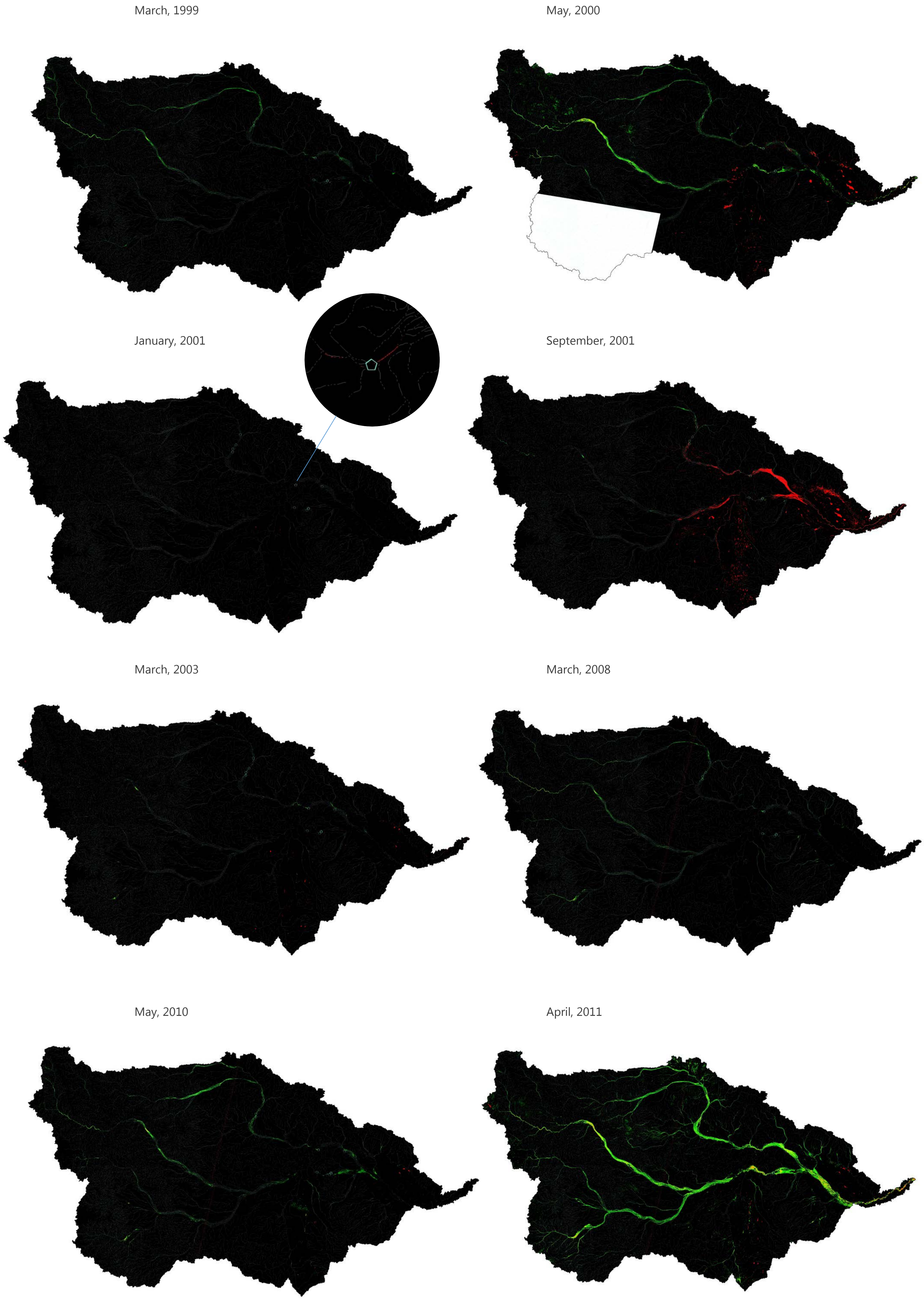


Figure 3-11: Catchment-scale hydrological variability in the Neales–Peake catchment (green to orange are NDVI 0.02–0.5; red is NDWI >0)

3.4.2.2 Representative reaches, Neales–Peake catchment

(a) Annual flows

Average annual flows for the four focal reaches are characterised by high inter-annual variability (CV >1), modelled at 0–894 GL in 1999–2013 (Table 3-4, Figure 3-12). Local catchments contribute an average 22–34% of annual flows. Under high-flow conditions, local contributions tend to be lower (Figure 3-13), consistent with local thunderstorms driving flows at small spatial scales (McMahon et al. 2008). The average seasonal flow variability is roughly bimodal, with peaks in late summer (40–80 GL/month) and winter (<40 GL/month). The winter peaks are shorter, from June to July, whereas there are extended flows from October, peaking in February to March. In the upper reaches (Arckaringa, Stewarts), the flow season peaks twice, in spring and summer, whereas the lower reaches (Algebuckina, Peake) peak once, in late summer. Other differences between reaches show similar trends among years and seasons (Figure 3-12). These comparisons should be viewed with caution, however, as there is uncertainty in modelled flow volumes, particularly at small scales (Montazeri and Osti 2014).

Table 3-4: Modelled annual flows (GL) at locations in the Neales–Peake catchment (1999–2013)

Reach	Scale*	Mean [GL]	CV**	Min [GL]	Max [GL]
Arckaringa Creek	all	90.8	1.27	1.22	449.3
@ d/s	local	29.2	1.16	0.00	127.9
Peake to confluence	all	204.3	1.16	0.03	894.0
@ Peake Crossing wh	local	54.0	1.19	0.00	215.1
Mathieson to Neales cnr	all	93.4	1.27	0.00	414.2
@ Stewarts wh	local	20.6	1.25	0.00	75.6
Algebuckina to Tard	all	107.7	1.26	0.00	467.4
@ Algebuckina wh	local	19.0	1.17	0.00	70.7

*Scale of influence was manipulated in Source IMS by excluding sub-catchments upstream of the location of interest (M Montazeri, DEWNR, pers. comm., 2014)

**CV = coefficient of variation (=standard deviation / mean); d/s = downstream; wh = waterhole

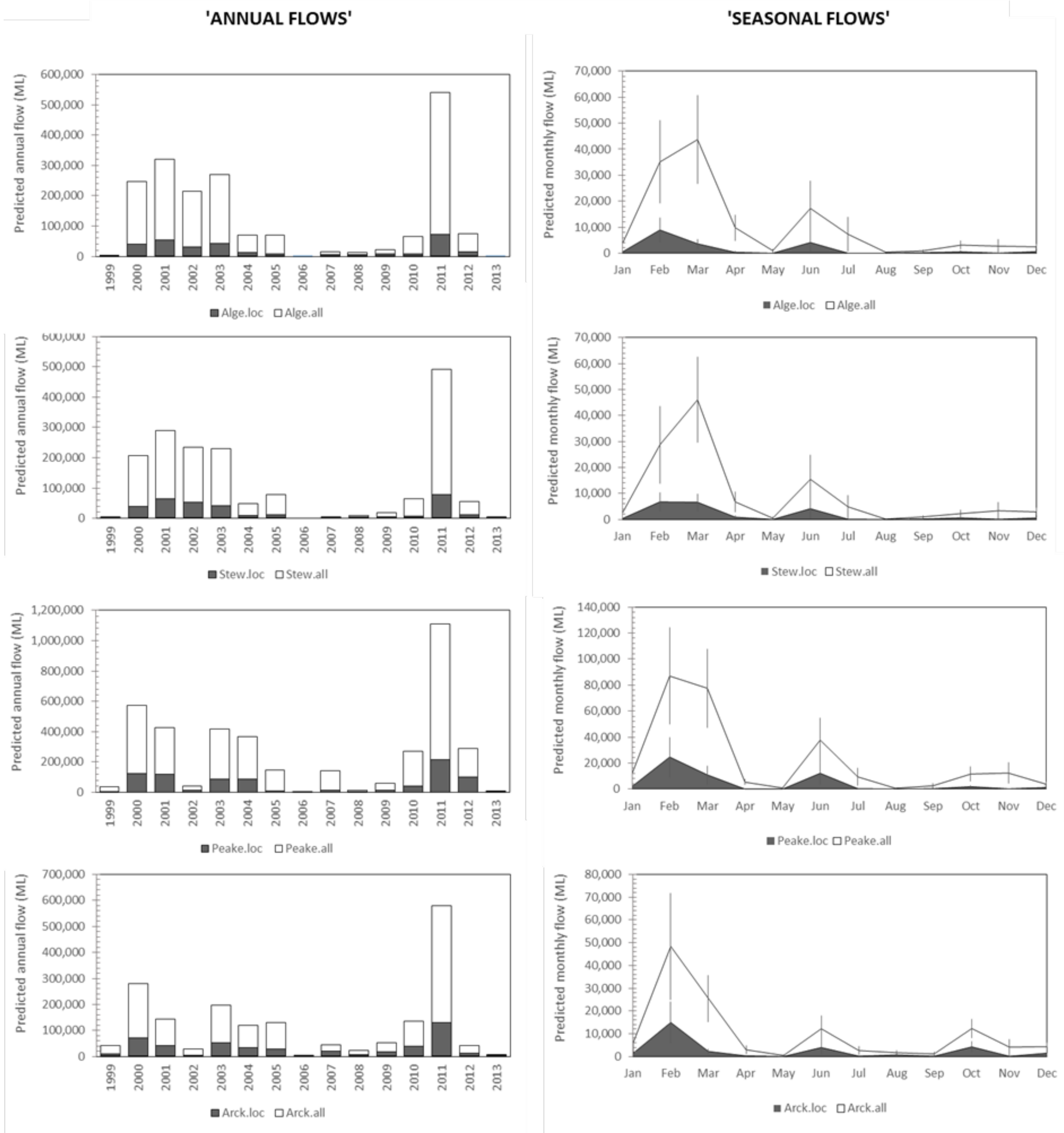


Figure 3-12: Predicted annual and seasonal flows supporting ecosystem indicators in the Neales–Peake catchment (seasonal data are mean ±SE)

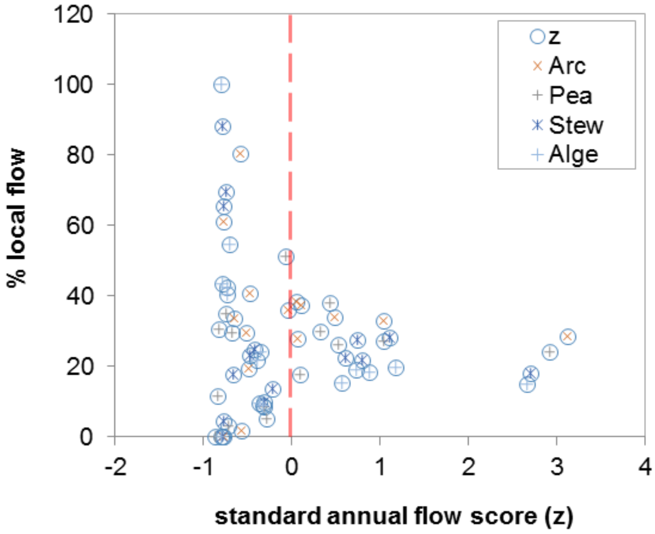


Figure 3-13: Relationship between standardized annual flow scores and percentage contributions of local catchments (dashed line = mean annual flow)

(b) Daily flows

Modelled hydrographs showing daily flows, water levels and incident-connecting flows for the representative reaches are in Figure 3-14. They show that flows that refill waterholes to or near cease-to-flow levels occur almost annually. Water levels in waterholes remain on a drying curve until the next flow, and only occasionally do they dry completely. An exception is the Arckaringa Creek reach, where water levels recede rapidly after flow events. The flow frequencies apparent in the hydrographs are surprising, given the intensely arid climate, but they concur with other observations (Costelloe 2011). The modelling suggests that in these rivers (compared to those in the eastern basin) there are lower flow durations and higher flow intermittency rather than differences in flow frequency or magnitude, coupled with higher potential evaporation (summer temperatures regularly exceed 45ºC) (McMahon et al. 2005). Flow durations in the WRLEB typically last for days rather than months, as in the eastern rivers. The modelled estimates show this in steep flow and no-flow duration curves (Qd and Zqd, Figure 3-15): flows exceeding 30 d are a 1:20 event, whereas no-flow days of 100 d or more occur every second or third year. These durations would constrain recruitment and dispersal of aquatic species and transfers of nutrients and water on the floodplain (Balcombe et al. 2007).

Linking connectivity to flow duration curves has been investigated using a simple flow transmission loss equation (Montazeri and Osti 2014), yielding a time-series (Figure 3-14) showing regular, intermittent connecting flows for all reaches. These flows occur in spells of >5 d once every 5 years or so (Qc in Figure 3-15). The estimates are consistent with satellite imagery indicating the infrequency of catchment-wide connectivity and free water on Neales–Peake floodplains (Figure 3-11).

Individual reaches showed subtle differences from these general patterns. Firstly, larger differences among reaches were observed in flow duration and intermittency rather than magnitudes. The upper reaches of the Neales and Peake had lower flow magnitudes and durations (30-d flows, representing a 1:20 year event) and high no-flow durations, so that connecting flows were rare (Figures 3-14, 3-15). In contrast, flow durations of >30 d were estimated in 1:3 and 1:5 years for Peake and Algebuckina, respectively, with extended flows connecting downstream refuges in the early 2000s and in 2010–11. This is consistent with the imagery in Figure 3-11. Secondly, differences in flow magnitudes between the upper and lower reaches were seen only in low-flow ranges, with low flows (<100 ML) showing a deviation in exceedance probabilities between upper reaches and Peake and Algebuckina (Figure 3-15). Subsequently, Algebuckina received regular, very low flows (<10 ML/day) and had much longer flow durations, associated with less severe dry runs and relatively stable water levels. These facets of the regime have implications for maintaining the hydroecological integrity of the Algebuckina refuge.

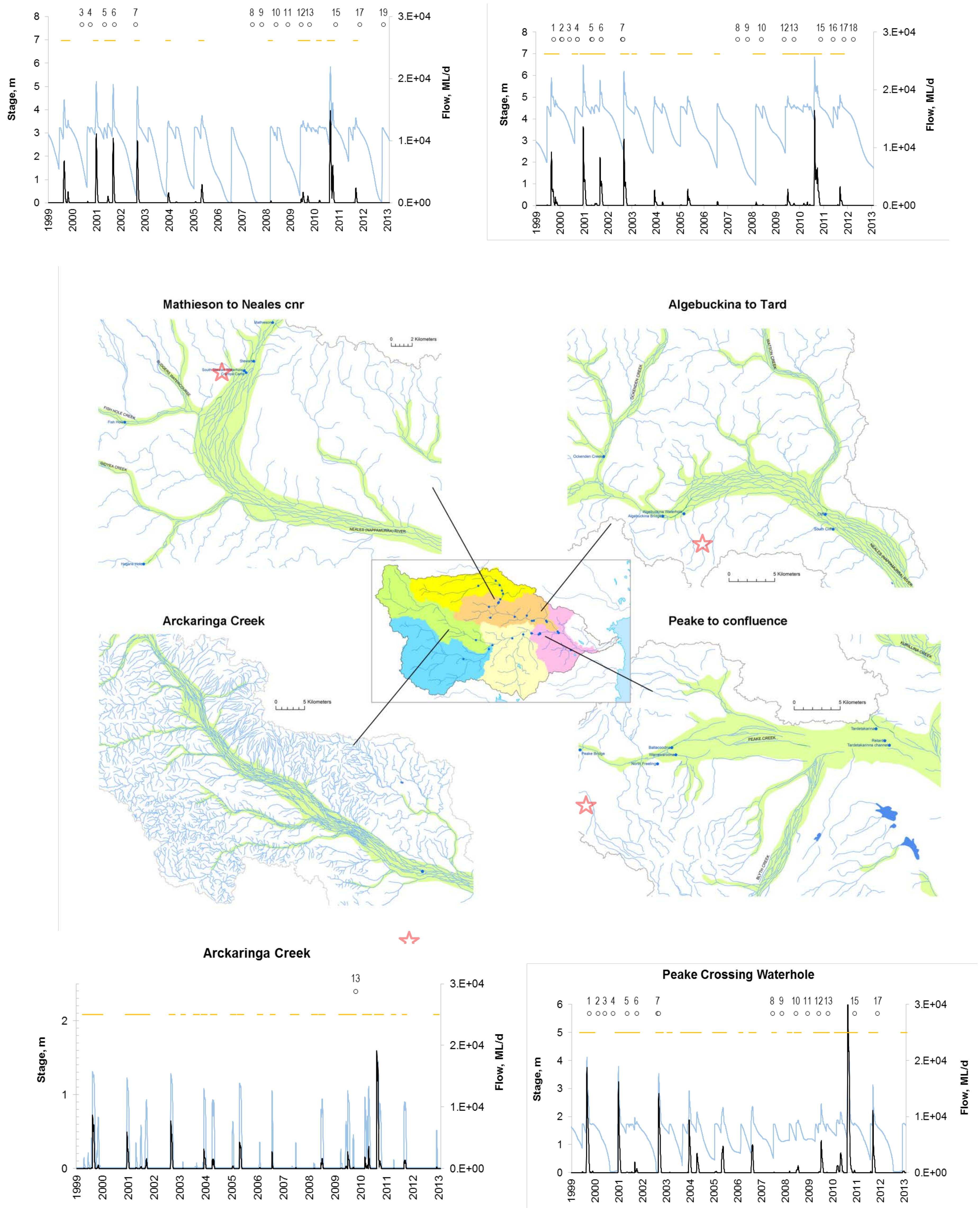


Figure 3-14: Hydrological time series in reaches of the Neales–Peake catchment (1999–2013)

*Modelled hydrograph locations are shown with red stars. The data show time series of modelled daily flows (black line), water levels (blue line) and incident-connecting flows (yellow line) in relation to fish monitoring (open circles).

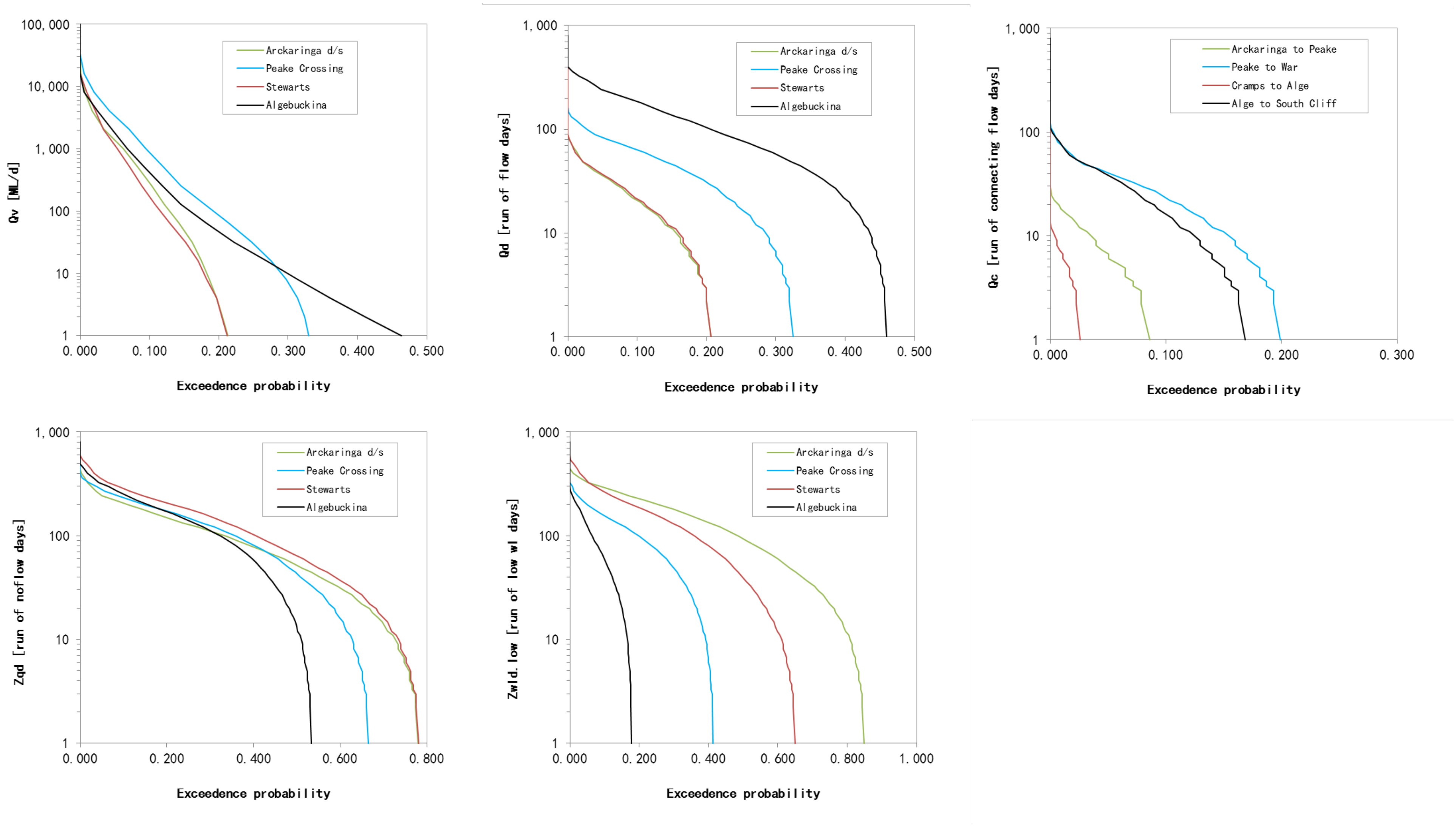


Figure 3-15: Modelled exceedance probabilities for flow magnitude (Qv), duration (Qd), connecting flows (Qc), no flows (Zqd) and low water levels (Zwld.low) in reaches of the Neales–Peake catchment

3.4.3 River salt balance

A feature of the mid-reaches of the Neales–Peake catchment is the interaction of flows with salinity. Given modelled predictions of a low incidence of connectivity between upper and lower reaches, salt accumulation is likely to increase during long no-flow periods, and may have caused the decline of Coolabahs along the mid-reaches of the Neales (Figure 3-17).

Moderate to large flow events (e.g. 1:20 to 1:100) connecting the upper and lower reaches may have a profound influence on salt transport (Costelloe et al. 2005b). The interaction is seen in the curvilinear relationship of salinity and flow (Figure 3-16)[10] over a broad temporal scale (90 d antecedent flow), relevant both to single large and multiple flow events. During no- or low-flow stages (Stages I and II_L) (0 < x < 2), salt is accumulated in downstream waterholes (but not in Stewarts, upstream of the saline mid-reaches of the Neales). After moderate to high magnitude flows (10-100 GL) (6 < x < 10) there is a sharp dip in salinity, representing dilution and flushing of salt from the river; at very high magnitude flows (>100 GL) (x > 10) there is a perceptible rise of salinity, probably related to small-scale temporal dynamics (short periods of maximal flushing of salt during the rising limb of the hydrograph). Costelloe et al. (2005b) demonstrated these relationships using modelled daily time series of flow and salinity.

Figure 3-16 indicates differences in base salt loads of the Neales (Stewarts, Algebuckina) and Peake, showing that these salinity-flow processes probably affect downstream refuges of the Peake only under natural flow conditions. Incidentally, modelled flows indicate that both the Peake and Arckaringa may receive higher magnitude flows more often, and may be more efficient in redistributing salt downstream. The difference in floodplain vegetation condition, with generally better conditions on the mid-Peake, supports this hypothesis. Additional factors include potentially higher inputs of salt from the Nilpinna and Freeling springs complexes *via* tributary flows, and flooding that may have a greater bearing on salt balance in the Peake-Warrawaroona waterhole. Salt balance was not modelled in the most recent studies due to insufficient data (Montazeri and Osti 2014), and these factors require more study (cf. Costelloe et al. 2005b).

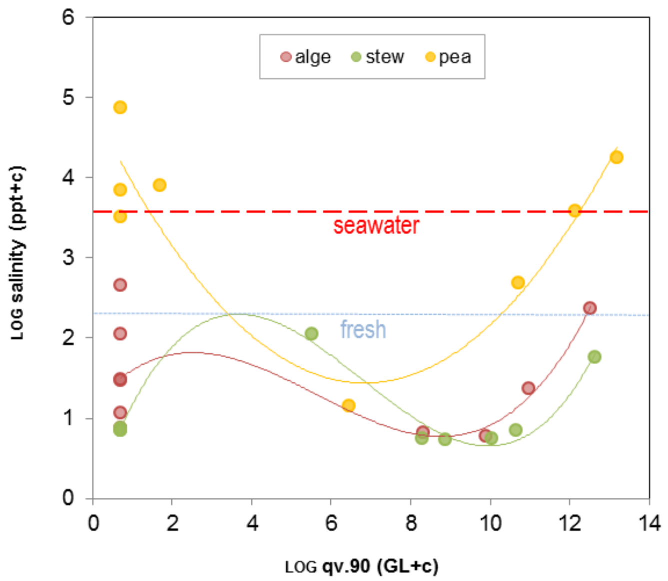


Figure 3-16: Curvilinear relationships of average salinity and 90 d antecedent flow (qv.90) (alge = Algebuckina; stew = Stewarts; pea = Peake Crossing)

[10] *On a log-log scale, salinity Y was predicted by qv.90 from: Algebuckina: Y = 0.0086x³-0.1459x²+0.5709x+1.1681; r²=0.39; Stewarts: Y = 0.0132x³-0.2686x²+1.4371+0.0092; r²=0.96; Peake Crossing: Y = 0.073x²-0.9996x+4.859; r²=0.83*

Figure 3-17: Vegetation in saline mid-reaches of the Neales and Peake rivers (right photo: JF Costelloe)

3.4.4 Environmental drivers: fish

3.4.4.1 Assemblage groups

A basin-wide analysis clearly shows the distinctiveness of the western rivers fish fauna, based on the dominance of hardy species and species endemic to the Finke catchment. The assemblage groups averaged 5.6 species per site and visit. To ease discussion here, species are referred to by their common names (for binomial names, see Table 3-5).

The assemblages fall into five groups related to catchments and refuge types, based on presence-absence of indicator species (Table 3-5). Table 3-7 shows the likelihoods of groups occurring in different refuge types and at different flow stages. The groups are characterised as follows (refuge types are explained in Section 2.2):

- **Group 1** includes ephemeral habitats (Disco refuges) of the western rivers, distinguished by species with high dispersal potential (periodic strategists) (Spangled Grunter, Bony Herring) (Table 3-5). The sample included the Neales and upper Macumba and one Finke waterhole (Pioneer Creek) under high-flow conditions.
- **Group 2** is the largest group of mainly western river habitats. The group has a more diverse assemblage than Group 1, averaging 5.9 species per visit per site. Sites encompassed a variety of ecosystem types (Ark, Disco, Polo Club refuges). All samples from Algebuckina Waterhole (Ark refuge) are in this group.
- **Group 3** includes all eastern river habitats as well as lower Macumba sites (Andarranna, Winkies). The group is distinguished by the presence of alien species (Goldfish) not found in WRLEB catchments and eastern river endemics (e.g. Carp Gudgeon species, Silver Tandan, Cooper Catfish, Australian Smelt). It has the highest richness, averaging 8.8 species per visit.
- **Group 4** includes Polo Club refuges of the western rivers and one sample from a Diamantina Disco refuge (Cowarie Crossing) during drought. It has the lowest richness of the groups and is distinguished by species with high salinity- and drought-tolerance (Desert Goby, Lake Eyre Hardyhead).
- **Group 5** includes all Finke habitats (except a Pioneer Creek sample in Group 1). It has the second highest richness, at 7.4 species per visit, and includes three endemics (Finke Purple-Spotted Gudgeon, Finke Goby, Finke Hardyhead).

Figure 3-18 illustrates the dissimilarity between the five groups, by group, catchment and site, as well as indicator species. The Polo Club sites with Group 4 indicator species (Desert Goby, Lake Eyre Hardyhead) in the Diamantina and Neales catchments are distinct from the refuges identified by Group 3 species, and Finke catchment sites identified by the endemic species of Group 5 are again distinct.

Ordination plots show species assemblage groups (1-5) based on presence-absence data for 30 waterholes in four catchments from 2007–12 (Appendix A). Each number assigns a site visit to a group based on its dissimilarity to other site visits (altGower index), where closer points are more alike in composition. Adjacent plots overlay indicator species, catchments and sites. For species abbreviations see Table 3-5; for catchment abbreviations see Table 3-7.

Table 3-5: Indicator species in fish assemblage groups. Non-waterhole refuge indicators are in parentheses (Sp=springs or bore drains; Dm=farm dams)

Species	Common	Abbr.	Group	indval	*P*
Leiopotherapon unicolor	Spangled Grunter	Lei.uni	1	0.254	0.011
Nematalosa erebi	Bony Herring	Nem.ere	1	0.231	0.001
			(Dm	0.596	0.12)
*Gambusia holbrooki**	Eastern Gambusia	Gam.hol	2	0.470	0.001
			(Sp	0.875	0.003)
Melanotaenia splendida tatei	Desert Rainbowfish	Mel.spl	2	0.287	0.001
Bidyanus welchi	Welch's Grunter	Bid.wel	3	0.949	0.001
Hypseleotris spp.	Carp Gudgeon	Hyp.spp	3	0.539	0.001
Macquaria ambigua	Golden Perch	Mac.amb	3	0.522	0.001
Porochilus argenteus	Silver Tandan	Por.arg	3	0.445	0.001
Scortum barcoo	Barcoo Grunter	Sco.bar	3	0.306	0.003
*Carassius auratus**	Goldfish	Car.aur	3	0.294	0.001
Retropinna semoni	Australian Smelt	Ret.sem	3	0.211	0.014
Neosiluroides cooperensis	Cooper Catfish	Neo.coo	3	0.059	ns
Chlamydogobius eremius	Desert Goby	Chl.ere	4	0.725	0.001
			(Sp	0.875	0.001)
Craterocephalus eyresii	Lake Eyre Hardyhead	Cra.eyr	4	0.558	0.001
Mogurnda larapintae	Finke Purple-Spotted Gudgeon	Mog.lar	5	0.800	0.001
Ambassis mulleri	Desert Glass Fish	Amb.mul	5	0.586	0.001
Neosilurus hyrtlii	Hyrtl's Catfish	Neo.hyr	5	0.540	0.001
Amniataba percoides	Barred Grunter	Amn.per	5	0.504	0.001
Chlamydogobius japalpa	Finke Goby	Chl.jap	5	0.300	0.002
Craterocephalus centralis	Finke Hardyhead	Cra.cen	5	0.300	0.004

*Introduced (alien) species

Table 3-6: Species richness of fish assemblages by catchment, refuge type and group

Catchment (abbr.)	***n***	**Mean**	**SD**	**SE**
Cooper (Coo)	16	8.38	2.09	0.52
Diamantina (Dia)	4	7.50	3.00	1.50
Finke (Fin)	11	7.09	1.30	0.39
Macumba (Mac)	12	4.42	2.47	0.71
Neales (Nea)	55	4.60	1.82	0.25
Refuge class	***n***	**Mean**	**SD**	**SE**
Ark	25	7.48	2.02	0.40
Disco	53	4.77	2.25	0.31
Polo Club	20	5.40	2.33	0.52
Assemblage group	***n***	**Mean**	**SD**	**SE**
1	25	3.12	0.88	0.18
2	37	5.92	1.46	0.24
3	17	8.82	1.85	0.45
4	9	3.00	0.87	0.29
5	10	7.40	0.84	0.27
Grand totals	**98**	**5.59**	**2.47**	**0.25**

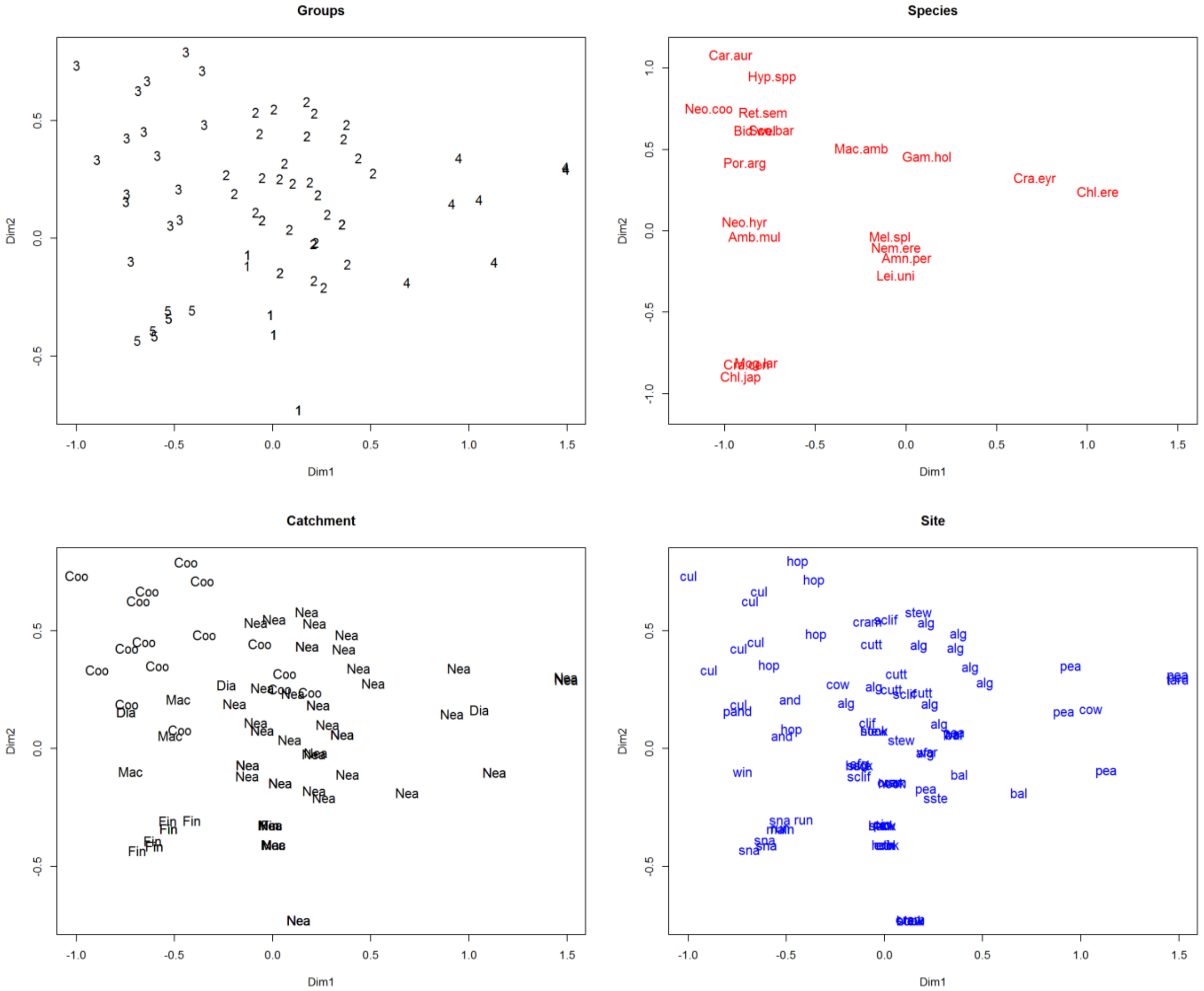


Figure 3-18: Ordinations of indicator species (top right) and fish assemblages by group, catchment and site

Table 3-7: Base likelihood of fish assemblage groups across refuge types and flow stages

Group	Refuge Type			Flow Stage*					
	Ark	*Disco*	*Polo Club*	*I*	*II_L*	*II_S*	*II_C*	*III*	*IV*
1	0.04	0.96	0.00	0.16	0.08	0.40	0.20	0.12	0.04
2	0.32	0.43	0.24	0.05	0.11	0.22	0.41	0.16	0.05
3	0.59	0.24	0.18						
4	0.00	0.11	0.89	0.11	0.00	0.44	0.11	0.11	0.22
5	0.30	0.70	0.00						

*Based on ARIDFLO models (Section 1.4.6). Flow states assigned by comparing hydrographs from a rainfall-runoff model (Montazeri and Osti 2014) among representative refuges for all sampling visits in the Neales–Peake catchment.

3.4.4.2 Eastern river refuges (Group 3)

Group 3 sites typically are Ark refuges (Table 3-7), with lower average temperatures and fresh to saline, well-oxygenated water. These habitats receive regular flows from the tablelands of Queensland, and remain recharged in most years. The magnitude and duration of flows are greater than in the western rivers, and include floods spreading over hundreds of square kilometres (Channel Country). Despite having a lower species richness than the Cooper and Diamantina, the lower Macumba assemblages include Group 3 species, consistent with the ability of many LEB species (apart from the rare, sedentary Cooper Catfish) to disperse widely under high-flow conditions (Kerezsy et al. 2013, 2014). Of the Group 3 indicator species, Welch's Grunter, Golden Perch and Silver Tandan occur in the Neales and Macumba catchments, but Silver Tandan has not been found in the Neales since the ARIDFLO surveys (2000–03). The ability of Group 3 assemblages to persist through drought is supported by permanent waterholes (Ark refuges, e.g. Cullyamurra) that provide littoral and deep-water habitats (Arthington et al. 2005).

3.4.4.3 Western river refuges (Groups 1, 2, 4)

Group 1 habitats typically are ephemeral aquatic habitats (Disco refuges) in the upper Macumba and Neales rivers. The group is characterized by highly mobile species ('periodic strategists'), Spangled Grunter and Bony Herring, which breed in local to catchment-wide floods (although Bony Herring may also recruit under no-flow conditions). The periodic breeding and recruitment behaviour of these species is shown in multiple peaks in length-size frequency histograms (Figure 3-20), although breeding in Bony Herring is more evenly spread over no flow-flow conditions, whereas Spangled Grunter breed almost in response to flow. These differences might be explained by different cues for breeding, with Spangled Grunter adapted to play a 'numbers game' of breeding and dispersing quickly as larvae or juveniles. These Disco refuge species could indicate alterations to natural flow variability in the western rivers.

Group 2 sites in the western rivers are most like eastern Group 4 sites, being deep, fresh and well-oxygenated with lower average temperatures (Figure 3-19). The assemblages support the highest species richness for western refuges (Table 3-6). While other western site assemblages may shift between Groups 1, 2 and 4, the assemblage in Algebuckina Waterhole (Ark refuge) is stable. Other refuge types supporting Group 2 assemblages are likely to receive significant flows (Table 3-7), enabling movements of species along reaches connected to Algebuckina or *via* Disco refuges that support breeding populations (McNeil et al. 2011). Two species associated with Group 3, Welch's Grunter and Golden Perch, also occur in deeper waterholes of the western rivers (>3 m cease-to-flow (ctf) depth: Appendix A), in habitats similar to those in the eastern waterholes. Golden Perch occur in very low numbers following extended drought in the Neales River, and often are undetected in surveys (Schmarr et al. 2014). This highlights the importance of Algebuckina for the western rivers fauna and the vulnerability of shared indicator species (Golden Perch, Welch's Grunter) to changes in the flow regime (cf. Cockayne et al. 2015).

Eastern Gambusia, an alien species that favours slow-flowing or still waters and a widespread pest in temperate rivers, is not favoured by variable flows. Small populations are likely to be displaced from smaller waterholes by flow pulses, limiting their spread in the western rivers. The species does persist, however, perhaps finding shelter in large waterbodies such as Algebuckina and bore drains and spring-fed wetlands that are occasionally connected to the river network (McNeil et al. 2011). Another indicator species, Desert Rainbowfish, also prefers sheltered habitats for spawning and recruitment (Arthington and Balcombe 2011). Although this is a ubiquitous, highly mobile species in the western river waterholes, recruitment may depend on sheltered habitats and floodplains in the Algebuckina reach. Both indicator species have low juvenile survivorship and low fecundity (Arthington and Balcombe 2011). Unimodal length-size frequency histograms for the Desert Rainbowfish (Figure 3-20) indicate sustained low recruitment regardless of flow, typical of short-lived, small species (McNeil et al. 2011). Both species are indicators of permanent, deep water conditions.

Group 4 assemblages typically occupy Polo Club refuges (less commonly Disco refuges). These habitats are shallow and prone to drying, saline (often more saline than seawater: 35 ppt) and subject to high temperatures (Figure 3-19). The Desert Goby and Lake Eyre Hardyhead are tolerant of salinity and hypoxia, and are effective indicators of developing saline conditions during dry spells in the lower Neales and Peake. The Desert Goby spawns on the underside of rocks or other coarse debris. Lake Eyre Hardyhead breed in contracted waterbodies, shown in length-age frequency peaks under dry conditions (Figure 3-20); this may reflect a requirement of larvae and juveniles for a higher prey density (Humphries et al. 1999; Shiel et al. 2006). For Lake Eyre Hardhead, a no-flow flow sequence primes populations for dispersal and recruitment, consistent with the species' occurrence throughout the western rivers. The Lake Eyre Hardyhead is a singular indicator of the importance of no-flow flow variability and naturally saline waters.

The members of Groups 1 and 4 include small- to medium-bodied species (Spangled Grunter, Bony Herring, Desert Goby, Lake Eyre Hardyhead) that tolerate a range of temperatures and poor water quality, typical of drying, saline reaches and isolated pools. Alien Goldfish, present in eastern sites (Group 3), do not occur in the western rivers (Groups 1, 2, 4).

3.4.4.4 Northern river refuges (Group 5)

Group 5 occupies a well-defined group of waterhole refuges in the upper and mid-Finke River. The refuges have significant ecological value, with high species richness and endemism (Table 3-6). Their occurrence on the Finke is consistent with the lack of modern-day connections with Kati Thanda–Lake Eyre. Early naturalists suspected that the river did occasionally flow to Kati Thanda–Lake Eyre (Bonython and Mason 1953), but it is now believed not to have been connected for at least 1000 years (Unmack 2001). The habitats are in good condition (Duguid et al. 2005) and distinguished by low turbidity, consistent with coarse bedload, high nutrients (nitrate-N) and low dissolved oxygen (Figure 3-19).

3.4.4.5 Flow as a 'disorganizer'

Wide spatial and temporal fluctuations are typical of fish assemblages in the Lake Eyre Basin (Arthington et al. 2005; Kerezsy 2010). The variable flow regimes drive the structure of assemblages by influencing breeding and recruitment events, movement between patches and population persistence, and variability can be thought of as an ecosystem driver or 'organizer' (cf. Walker et al. 1995). Flow variability within and between years is so pronounced that communities are in a constant state of flux, and flow might equally be called a *disorganizing* variable in these systems. Its effect is to reset the composition of assemblages.

Environmental surfaces of principal flow axes (Principal Components: PCs), representing a natural flow regime for the western rivers (Figure 3-21), indicate the presence/absence of species in different habitats under different flow conditions. Facets of the flow regime include flow stage (PC1), flow intermittency (PC2), water-level fluctuation (PC3) and time since inundation (PC4), together explaining 73% of variation in modelled variability between sampling times (Appendix D.2). The relationships of these flow facets to the presence/absence of indicator species illustrate the 'disorganizing' influence of flow on the fish.

Under flow-pulse conditions (positive PC1 scores; Figure 3-21), assemblages are temporarily homogenized as indicator species are dispersed (or dislodged) from their refuges, and few are detected during flow stages. Species detected most often during flow stages (Lake Eyre Hardyhead, Bony Herring, Desert Rainbowfish) are those which actively recruit in response to flow (Figure 3-20). In contrast, under no-flow conditions, more species regain their preferred habitats and are detected in surveys. This is consistent with a lag time for re-establishment of benthic algae, supporting food webs in ephemeral stream habitats (Bunn et al. 2003; Arthington and Balcombe 2011). The positive influence of a sufficient lag time is seen in the relationship between time since inundation (negative PC4 scores, Figure 3-21) and the presence of flood-dependent species (Spangled Grunter, Bony Herring) and species utilizing floodplain habitats for spawning and recruitment (Desert Rainbowfish). Although Golden Perch utilize extensive floodplain river habitats in the eastern basin, the species' positive score along this axis suggests that floods in the western rivers may be too short-lived or too shallow for this large-bodied species to recruit effectively (cf. Cockayne et al. 2015).

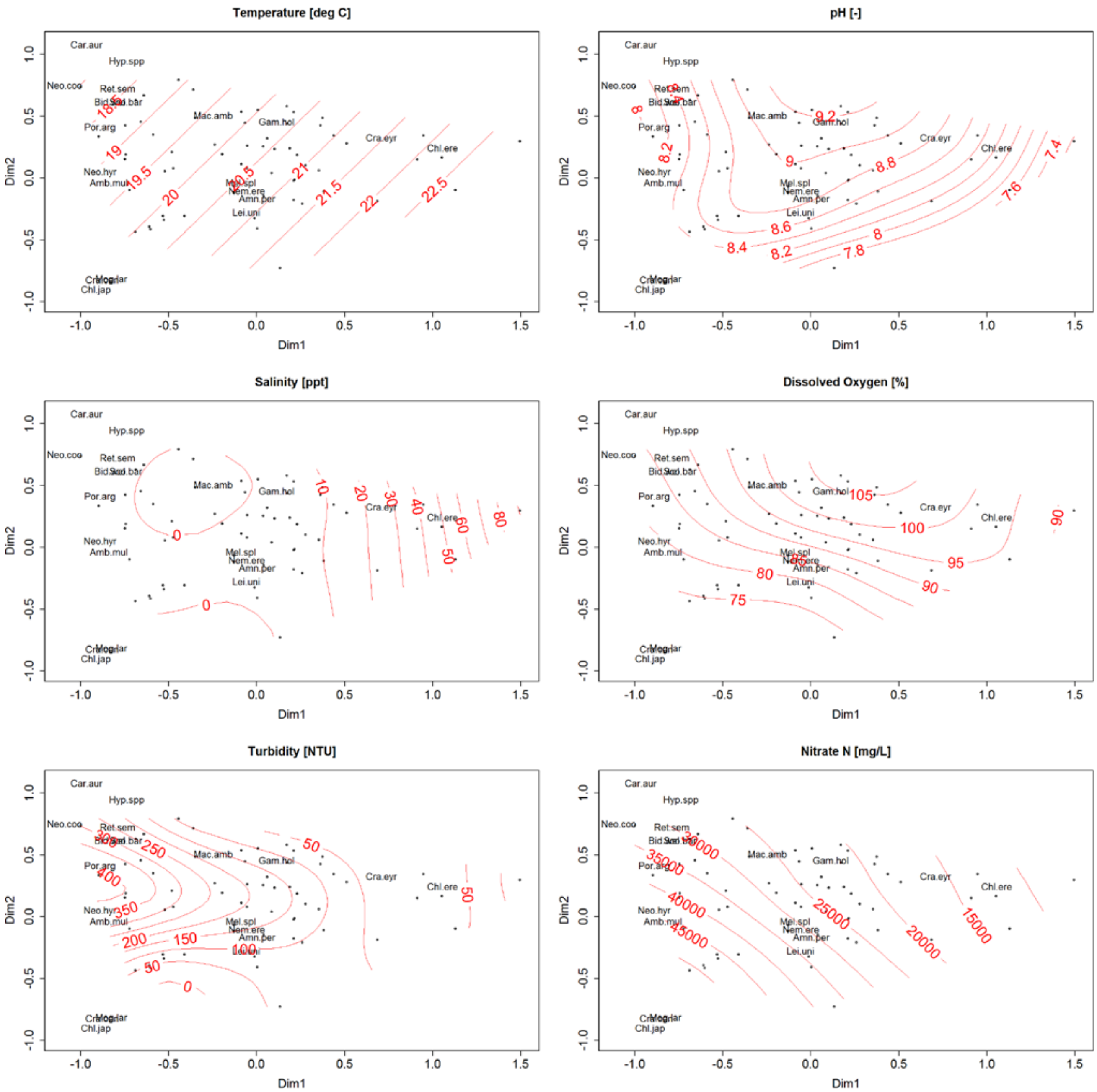


Figure 3-19: Water-quality drivers of fish assemblages and indicator species[11]

[11]*Plots show isohyets of water quality variables significantly related to the five groups. Isohyets were generated using a Generalized Additive Model (see Methods). For species abbreviations, see Table 3-5.*

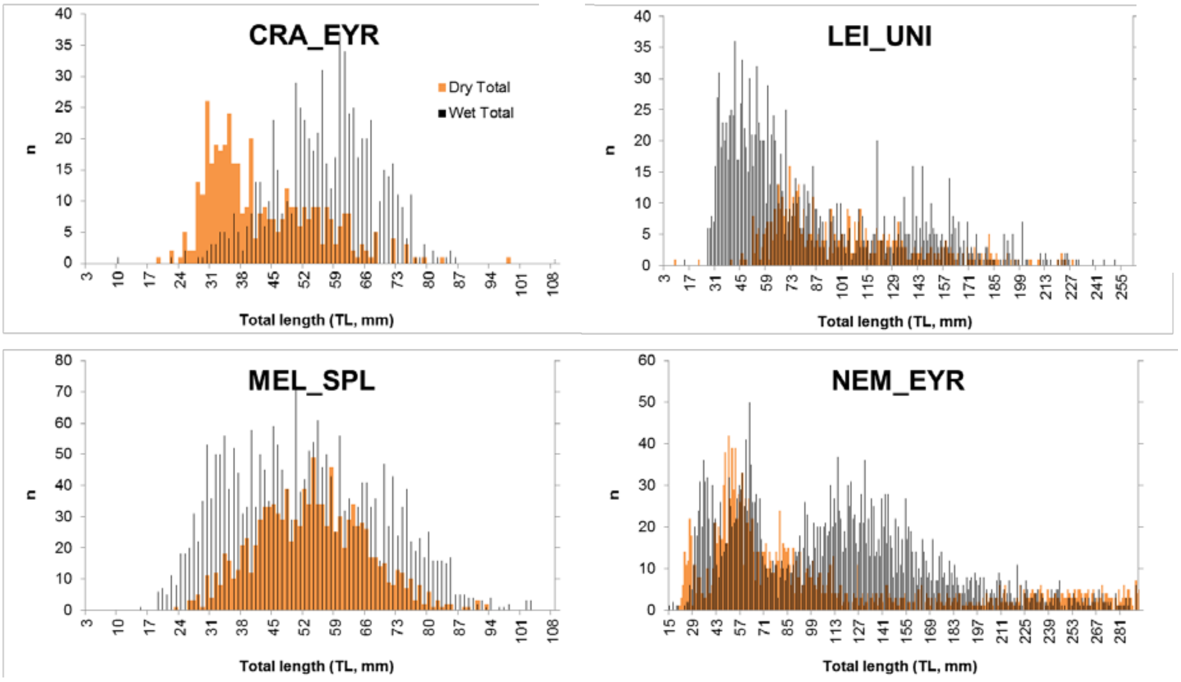


Figure 3-20: Length-size frequency histograms for species under wet (grey: flow, Stages II–III) and dry conditions (brown; no flow, Stages I, IV)

CRA_EYR = *Craterocephalus eyresii* (Lake Eyre Hardyhead), LEI_UNI = *Leiopotherapon unicolor* (Spangled Grunter), MEL_SPL = *Melanotaenia splendida tatei* (Desert Rainbowfish), NEM_EYR = *Nematalosa erebi* (Bony Herring). Peaks below the minimum spawning age-length class are breeding events and peaks above the mean spawning age-length class are recruitment events (see Table 3-8). These are pooled data peaks and should be considered as modal classes (i.e. they are not event-specific).

Table 3-8: Spawning size class data for LEB fish species (Total Length, TL)

Genus	Species	*n*	Mean [mm TL]	CV	Min [mm TL]	Max [mm TL]	Range [mm TL]
Amniataba	*percoides*	60	76.9	0.26	54	133	79
Chlamydogobius	*eremius*	18	47.7	0.10	42	57	15
Chlamydogobius	*japalpa*	4	44.5	0.08	41	49	8
Craterocephalus	*centralis*	26	54.8	0.12	38	68	30
Craterocephalus	*eyresii*	212	54.2	0.25	28	85	57
Gambusia	*holbrooki*	17	37.1	0.27	27	64	37
Hypseleotris	spp.	9	31.2	0.16	27	42	15
Leiopotherapon	*unicolor*	176	101.3	0.30	51	211	181
Macquaria	sp.	3	323.7	0.22	266	402	136
Melanotaenia	*splendida tatei*	10	59.7	0.22	48	90	42
Nematalosa	*erebi*	22	219.5	0.23	128	315	187

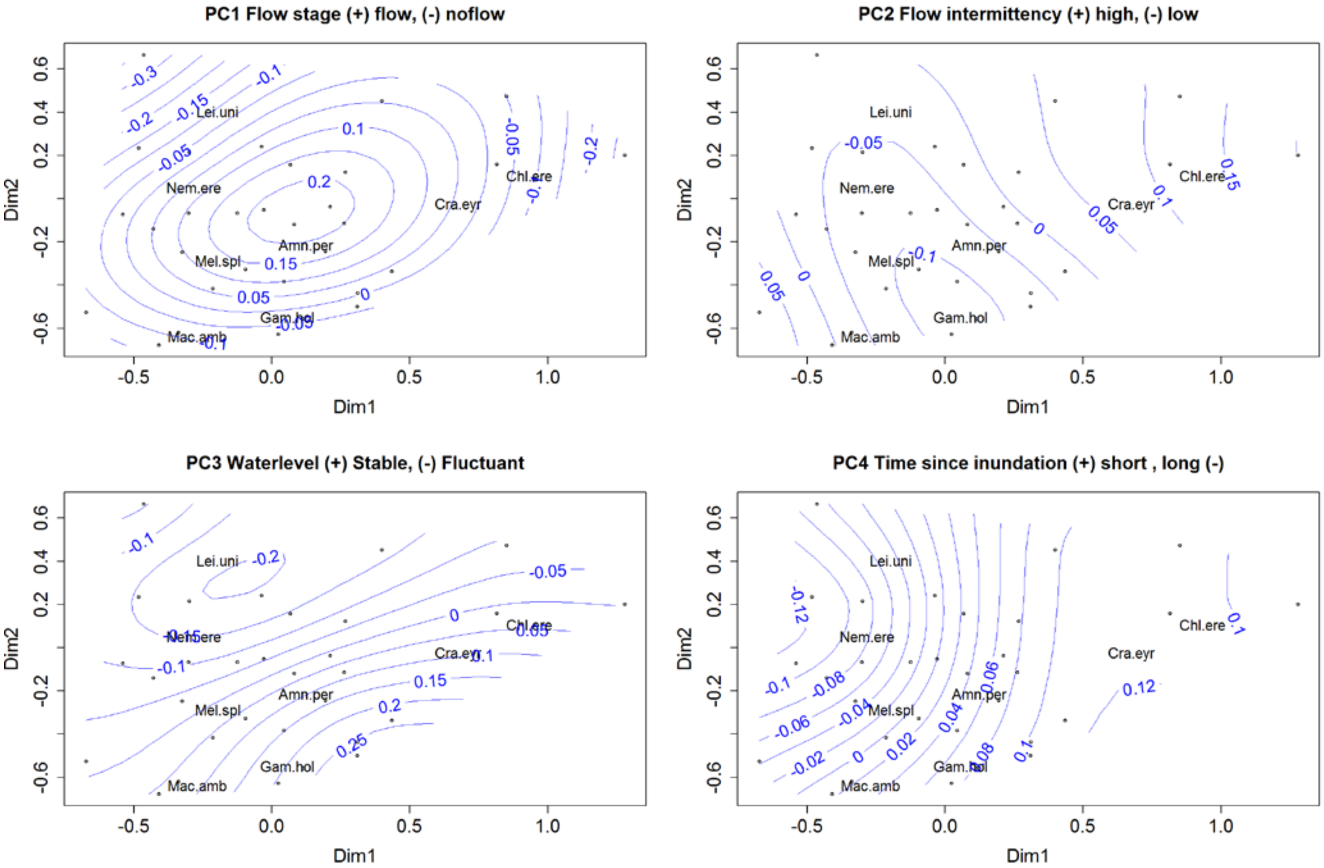


Figure 3-21: Fish indicator species in Neales refuges, related to principal axes of flow variation (PC1–PC4)

PC4 (bottom right) shows that Bony Herring (Nem.ere) and Spangled Grunter (Lei.uni) are more likely to occur where time since inundation is longer (negative PC4), while Lake Eyre Hardyhead (Cra.eyr) and Desert Goby (Chl.ere) occur where the time is shorter (positive PC4). For methods see Appendix D.2; for species abbreviations, see Table 3-5.

Periods of low flow intermittency (negative PC2 scores) favour species reliant on connecting flows for dispersal and recruitment (Spangled Grunter, Bony Herring). These conditions are associated with the presence of Eastern Gambusia as well as opportunists (Desert Rainbowfish, Barred Grunter) favoured by a more constant flow regime and occasionally detected in Algebuckina Waterhole (Section 3.4.4). Conversely, species requiring intermittent flows to maintain breeding and recruitment (Lake Eyre Hardyhead, Desert Goby) indicate an intermittent regime, characteristic of the Peake River and Disco refuges of the upper Neales River.

More permanent water levels (positive PC3 scores) favour more species, as in the diverse assemblage of Algebuckina Waterhole. The moderate environmental stressors in this refuge support breeding and recruitment of many species, particularly those with relatively low fecundity (Eastern Gambusia, Desert Rainbowfish, Golden Perch). Although the upper Neales and Peake waterholes occasionally dry, refuges of the lower Peake support salt-tolerant species (Lake Eyre Hardyhead, Desert Goby) adapted to intermittent flows. It is likely these populations maximize opportunities to reproduce under occasional, catchment-scale flows, as in 2010–11. Their association with moderately stable water levels ($0 < PC3 < 0.1$), when they are normally adapted to intermittent flows (PC2, PC4), demonstrates the influence of flows that disrupt assemblages in Ark, Disco and Polo Club refuges.

Pulsed river flows cause large shifts in fish abundance, in response to disturbance and food sources established during extended no-flow periods (cf. Arthington et al. 2005; Kerezsy 2010). Ordinations of scaled abundance (Figure 3-22) in Algebuckina, Cliffs and Stewarts, three refuges on the Neales River, demonstrate responses to the ever-changing hydrological conditions (Figure 3-23). Algebuckina (Ark refuge) shows some predictability between 'wet' and 'dry' condition assemblages, indicated by opportunists (Barred Grunter, Desert Rainbowfish, Eastern Gambusia) and tolerant specialists (Desert Goby, Lake Eyre Hardyhead), respectively. Conversely, large and unpredictable shifts occur in Disco waterholes, particularly in the upstream refuge (Stewarts). Large, pulsed flows cause the reappearance of cryptic species (e.g. Welch's Grunter, Golden Perch) and the disappearance of others, seen in shifts in assemblage structure (i.e. all sites shift towards [0,0] from visits 12-15). Similarly, severe drought results in convergence of sites towards a species-poor assemblage.

Only part of the variation in relative abundance can be attributed to natural hydrological conditions. Local flows may occur in individual reaches, allowing connection of waterholes to artificial waterbodies (dams, bore drains) that function as refuges for some species. Dams, in particular, may hold water for extended periods and typically support populations of Spangled Grunter and Bony Herring, although low habitat diversity and low dissolved oxygen may make them unsuitable for other species. Spangled Grunter and Bony Herring are the most common species of Disco waterholes, and the only species found during opportunistic sampling of in-channel habitats in the upper reaches (Lora, Arckaringa creeks) during flow events (e.g. high flows in 2010–11) (Figure 3-22). Bore-drain wetlands and springs, on the other hand, are refuges for Eastern Gambusia and small populations of salt-tolerant native species (Desert Goby, Lake Eyre Hardyhead). These species often appear suddenly in upper Neales Disco refuges and isolated reaches, suggesting that upper and lower reach populations are mostly disconnected; instead, they may disperse around the refuges. McNeil et al. (2011) proposed that dams function similarly to Disco refuges, and bore-drain wetlands to spring wetlands. This concurs with modelling suggesting only intermittent connections (days, rarely weeks) between upper and lower Neales reaches under very high flow conditions (Montazeri and Osti 2014). These may not be suitable timeframes or conditions for dispersal of some small-bodied species.

There is a problem in analysing fish abundances, due to problems of detecting cryptic species and the disorganising influence of flows. Although a method of scaling was trialled at the waterhole scale (Table 3-9), the abundance data appear random with clear evidence of false negatives (zeros). Although statistical techniques are available to handle these kinds of data, without estimates of detectability it is difficult to determine a best approach. Experiments could be trialled to quantify detectability (under different flow and no-flow conditions); this would provide guidance for monitoring designs and a more robust platform for use of abundance data in hydroecological models.

Table 3-9: Scaled fish abundances in three waterholes of the Neales River (see also Figure 3-23)

Site visit	Hydrological conditions*			Species abundance†								
	qd.90	*qv.90*	*zqd.spell*	*Amn.per*	*Bid.wel*	*Chl.ere*	*Cra.eyr*	*Gam.hol*	*Lei.uni*	*Mac.amb*	*Mel.spl*	*Nem.ere*
alg.08	0	0	297	9	0	4	0	53	0	0	200	0
alg.09	0	0	443	0	0	0	1	0	0	0	6	0
alg.10	29	4	61	0	0	9	23	0	0	0	23	0
alg.11	0	0	151	0	0	0	13	0	0	2	0	0
alg.12	0	0	321	133	0	0	0	7	0	0	92	0
alg.13	71	20	0	0	0	0	0	1	0	0	227	0
alg.15	83	279	7	5	<1	0	0	0	41	40	0	0
alg.16	0	0	195	0	0	6	0	0	0	0	712	0
alg.17	60	58	16	0	0	0	<1	0	0	0	4	0
alg.18	0	0	161	0	0	4	0	0	0	1	0	0
alg.19	0	0	384	<1	0	0	0	<1	0	0	0	58
sclif.12	0	0	271	0	0	0	0	0	<1	0	0	2
sclif.13	90	27	0	0	0	0	0	0	0	<1	0	0
sclif.19	0	0	318	0	0	0	0	<1	0	<1	0	0
stew.10	22	4	68	0	0	0	0	0	<1	0	0	0
stew.11	0	0	140	0	0	0	<1	0	0	0	<1	0
stew.12	23	7	0	0	0	0	0	0	0	0	0	<1
stew.13	68	23	2	0	0	0	0	0	0	0	1	0
stew.15	76	311	14	0	0	0	0	<1	0	<1	0	<1
stew.17	49	43	28	0	0	0	0	0	<1	0	0	0
stew.19	13	0	17	0	0	0	0	<1	0	0	0	0

*Modelled conditions at time of visit based on 90-d antecedent flow (qd.90), magnitude (qv.90 GL) and length of dry spell (zqd.spell days)

†Abundances scaled according to waterbody size from calibrated water-level data. For species abbreviations, see Table 3-5.

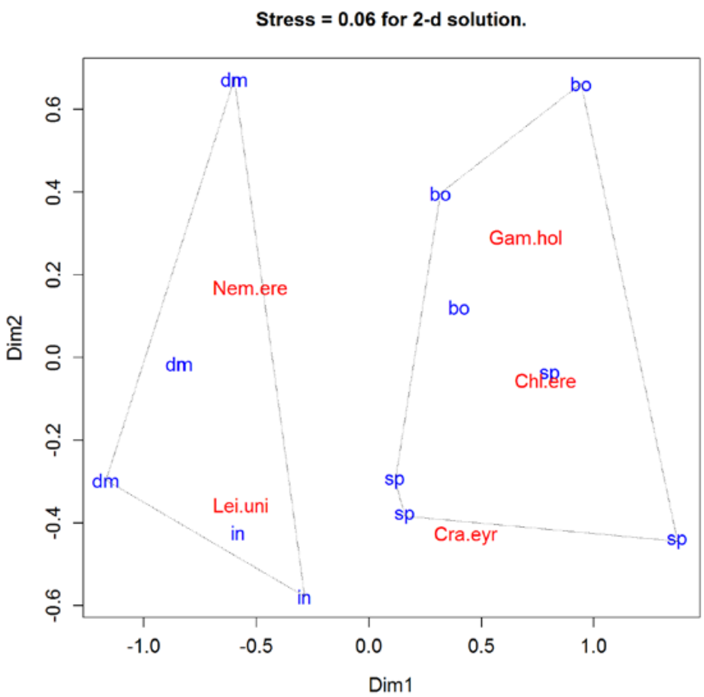


Figure 3-22: Ordination of fish assemblages in non-waterhole refuges and in-channel refuges

Lines are the boundaries of two groups, one represented by dams (dm) and in-channel habitats (in) and the other by bore-drain wetlands (bo) and springs (sp). For species abbreviations, see Table 3-5.

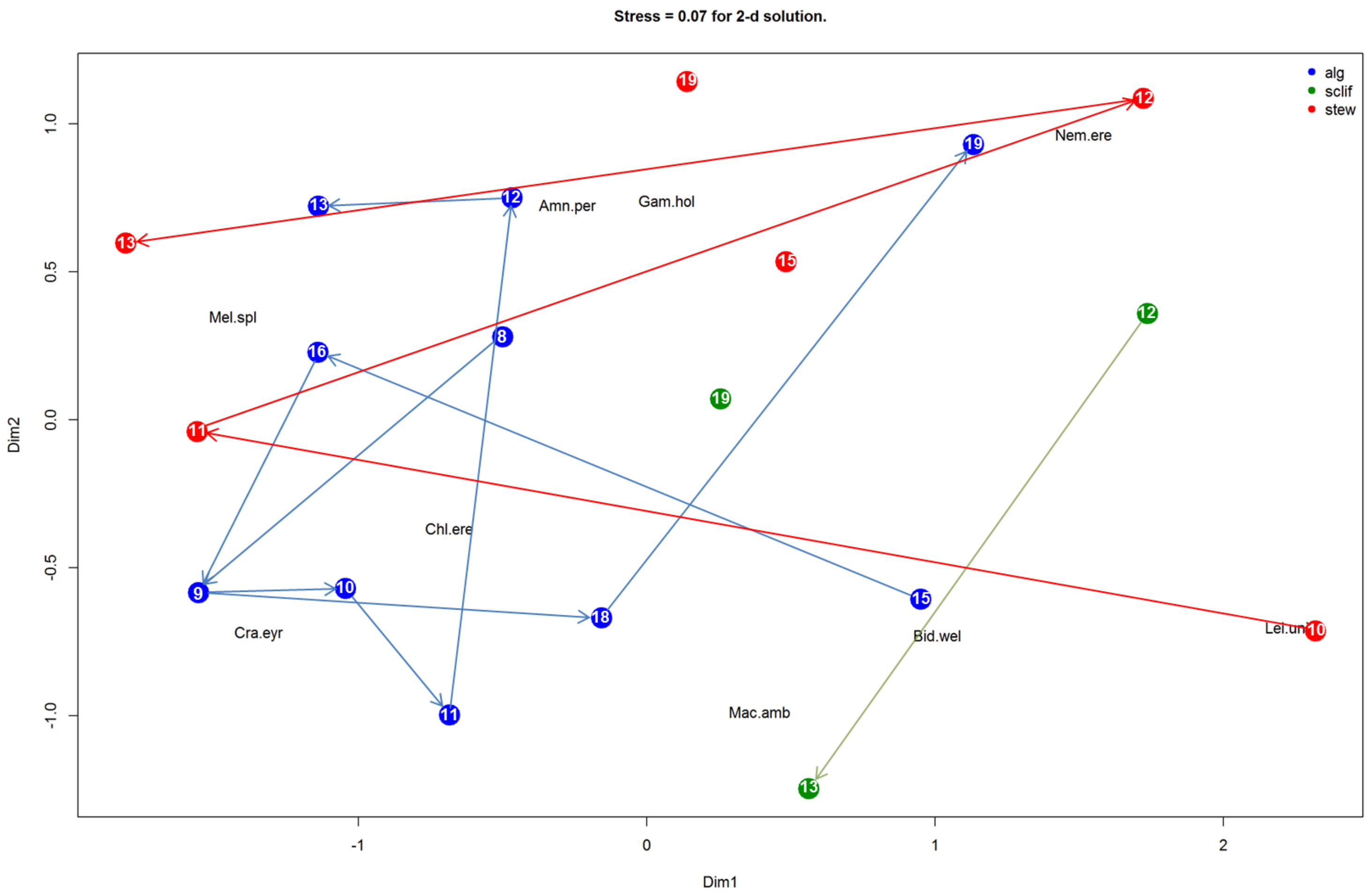


Figure 3-23: Ordination of scaled fish abundances for Algebuckina, Cliffs and Stewarts, 2007–13

Dots represent a scaled relative abundance estimate of assemblages at each site visit, with adjacent dots being most similar to each other (Bray-Curtis index). Numbers in circles refer to visits (Figure 3-14). Arrows represent time; no arrow indicates a visit was missed. Abundance for each species at site s and visit v was scaled by a factor 'wh.st', calculated as: wh.v s,v / wh.vmax , where wh.v is the waterhole volume (GL) estimated from bathymetric data (Montazeri and Osti 2014).

3.4.5 Environmental drivers: vegetation

3.4.5.1 Assemblage groups

The western floodplains support a mosaic of vegetation governed by patch- and landscape-scale drivers. Flood history, habitat conditions and present flow regimes shape the assemblages (cf. Friedel et al. 1993), seen in the imagery (Figures 3-9 to 3-11) as a patchwork of expanding and contracting areas of greenness. These factors are analysed to identify indicators and response types of assemblages that could reflect flows affected by mining and CSG development (Section 3.4.5). The Arckaringa subregion (Neales–Peake catchment) is again a focus, but other data from the western rivers are included for landscape-scale comparisons.

Survey sites on the floodplain and in flood-prone areas (Group 1, Appendix D.4) were analysed separately. This revealed four reasonably distinct groups, shown in Figure 3-24 (R = 0.73, P = 0.001), based on the relative abundances of indicator species (Table 6-10). The indicators for floodplains, floodouts and drainage lines occur across several communities identified by Brandle (1998), representing overstorey and understorey plants of the dominant vegetation-landform units, namely floodplains, claypans, stony gilgais and sandplains. This is typical of desert floodplain assemblages (Brandle 1998: p. 49):

> *Drainage channels are often a denser version of the surrounding (dry vegetation), which may develop into woodlands (wattle, Coolabah and occasionally River Red Gum) which feed into floodouts, swamps and pans.*

Indicator Species Analysis is a summation of groups of species that occur more frequently together than apart. In the following, common names only are used (for binomial names, see Table 3-10). From Table 6-10, flooding regime and soil properties tend to be shared for these species:

- **Group 1** species occupy periodically-flooded habitats such as Cane Grass swamps and claypans that occupy microrelief (i.e. low lying areas or depressions). Where there are clay-based soils, infiltration of rainfall or surface water is low and water may be ponded. Some of these habitats are disconnected from the modern floodplain and are the result of alluvial deposition during historic mega-floods (Pickup 1991); others are connected only during floods that inundate wider areas of the floodplain.
- **Group 2** species occupy frequently-flooded (1:5) areas connected to the river or to run-on areas that receive regular local flows (e.g. waterholes). Key species are woodland overstorey and understorey plants, Lignum, Coolabah and River Cooba, forming riparian corridors (and habitat for birds: Reid and Gillen 2013) along many of the western rivers. Group 2 species are drought-tolerant and moderately salt-tolerant.
- **Group 3** species occur in frequently-flooded (1:2) areas connected to the riverine environment or to run-on areas that receive regular local flows (e.g. waterholes, some inter-dunal wetlands). They are most common in the northern bedload rivers and fringing inter-dunal wetlands of the Pedirka and Simpson deserts, but occur also in and around the western waterholes. Group 3 species prefer sandy soils and include the iconic River Red Gum and associated grasses (Silky Browntop, Kerosene Grass) along northern river waterholes and sandy river banks. The group also includes drought- and flood-tolerant species, including shrubs (e.g. Sandhill Wattle) and grasses (e.g. Sandhill Canegrass), typical of desert dunes.
- **Group 4** species occur in infrequently-flooded areas, often intergrading with Groups 2–3 in areas of higher relief or on the outer margins of the floodplain. Gidgee grows along the margins of low-lying floodplains and is dominant along upland rivers with stony, clay loam soils. Wide distributions and variable habitats are features of these species, growing in different landforms (and in rocky areas and stony gilgais) higher in the landscape. Many occupy pastoral habitats and are heavily utilized for fodder (e.g. Barley Mitchell Grass, Bladder Saltbush) and others occur where there has been sustained heavy grazing (e.g. Nitre Bush) (Cunningham et al. 2011).
- **Group Sp** includes wetlands associated with GAB springs and bore drains. The characteristic species are aquatic plants requiring regular, long periods of inundation. Bore-Drain Sedge and Common Reed form extensive stands under these conditions; the latter forms mono-specific stands in permanent and seasonal habitats across Australia (Roberts and Marston 2011).

Table 3-10: Indicator species of riparian vegetation and flood-dependent groups (wl: water level)

Species	abbr.	Common name	Group	indval†	P	Habitat‡
Teucrium racemosum	Teu.rac	Grey Germander	1	0.517	0.003	infrequent major flooding
Sclerolaena bicornis	Scl.bic	Goathead Burr[1]	1	0.504	0.012	variable alluvial flats
Chenopodium auricomum	Che.aur	Golden Goosefoot	1	0.452	0.009	above wl clay soils
Eragrostis dielsii	Era.die	Mulka[1]	1	0.303	0.015	calcareous sandy soils, short-lived perennial
Eragrostis australasica	Era.aus	Canegrass	1	0.207	0.081	intermittent flooding clay soils
Duma florulenta	Mue.flo	Lignum[1]	2	0.599	0.002	at wl
Eucalyptus coolabah	Euc.coo	Coolabah	2	0.418	0.004	above wl
Acacia stenophylla	Aca.ste	River Cooba, Eumong	2	0.278	0.047	at or above wl
Eucalyptus camaldulensis	Euc.cam	River Red Gum	3	0.918	0.001	at or above wl
Eulalia aurea	Eul.aur	Silky Browntop	3	0.503	0.001	variable usually moist areas w rg
Eriachne ovata	Eri.ova	Wanderrie Grass[1]	3	0.352	0.023	variable xero (rocky), meso (outwash plain) forms
Acacia ligulata	Aca.lig	Sandhill Wattle	3	0.333	0.008	lake edges sandy soils
Acacia salicina	Aca.sal	Cooba[3]	3	0.316	0.037	above wl, heavy soils
Aristida holathera	Ari.hol	Kerosene Grass	3	0.286	0.013	variable widespread, sands w cg
Zygochloa paradoxa	Zyg.par	Sandhill Canegrass[3]	3	0.253	0.035	sandy soils and dunes
Acacia cambagei	Aca.cam	Gidgee	4	0.899	0.001	above wl clay soils
Einadia nutans	Ein.nut	Climbing Saltbush	4	0.571	0.001	heavy soils
Enchylaena tomentosa	Enc.tom	Ruby Saltbush	4	0.500	0.004	common generalist
Maireana aphylla	Mai.aph	Cottonbush	4	0.497	0.002	variable alluvial flats w bladder saltbush
Eragrostis setifolia	Era.set	Neverfail[1]	4	0.408	0.011	variable periodically flooded
Atriplex spongiosa	Atr.spo	Pop Saltbush	4	0.312	0.036	variable, resp to short-term flooding
Frankenia sp.	Fra.sp.	Sea Heath	4	0.284	0.091	
Atriplex vesicaria	Atr.ves	Bladder Saltbush[1]	4	0.254	0.064	variable widespread, alluvial plains w mg
Astrebla pectinata	Ast.pec	Barley Mitchell Grass[1]	4	0.211	0.066	gilgai or alluvial sands, wet season increaser
Myoporum montanum	Myo.mon	Western Boobialla[3]	4	0.158	0.095	variable widespread, depressions
Nitraria billardierei	Nit.bil	Nitre Bush[3]	4	0.158	0.097	above wl clay soils, saline creek flat
Cyperus laevigatus	Cyp.lae	Bore-Drain Sedge[2]	Sp	0.97	0.001	below wl, in shallow waters of artesian bores
Phragmites australis	Phr.aus	Common Reed[2]	Sp	0.177	0.001	at wl, shallow intermittent floods or marshes

[1]fodder species; [2]drought fodder species (or part-grazed); [3]unsuitable for stock (or overgrazing increasers).

† Relative contribution to group structure based on indicator species analysis (Roberts 2013)

‡ Notes from Cunningham et al. (2011) and other sources (e.g. http://bie.ala.org.au/; www.worldwidewattle.com/)

[1]Syn. *Muehlenbeckia*

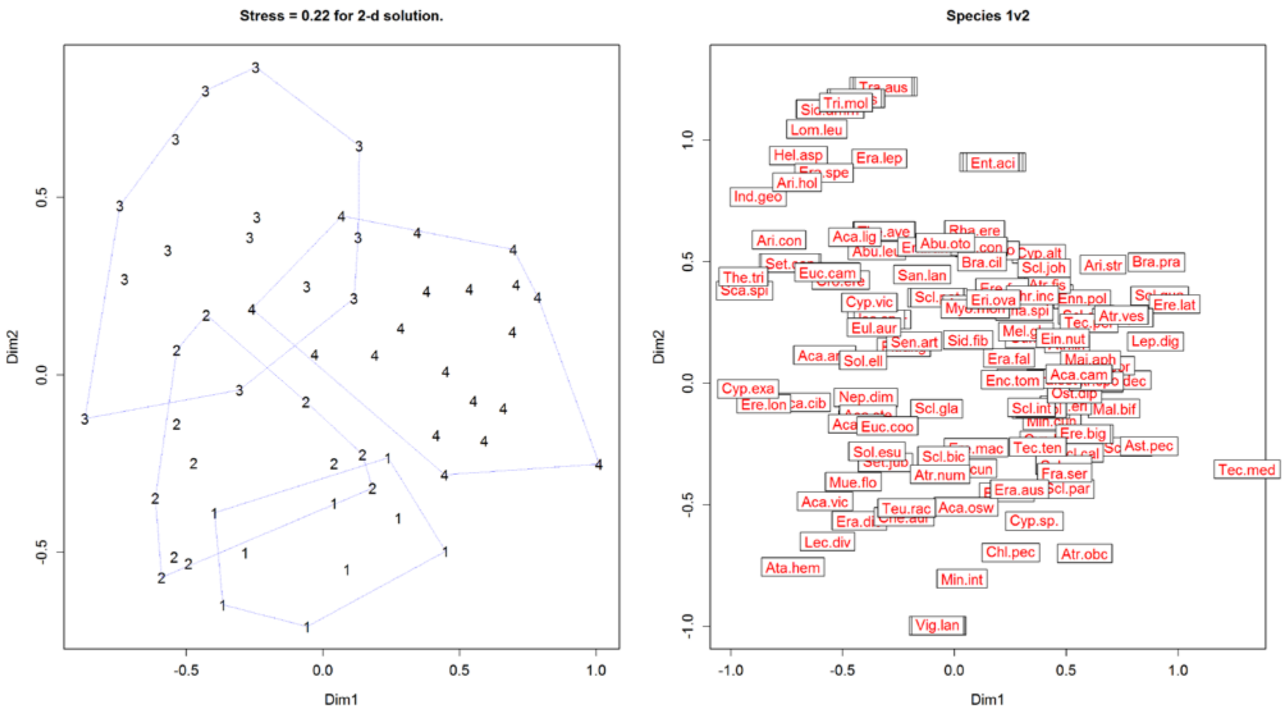


Figure 3-24: Riparian and flood-dependent plant groups[12]

The floodplain assemblages are structured on a hierarchy of scales, according to broad landscape-scale catena (or geology-landform relationships), patch-scale drainage patterns (or microrelief) and distribution of soils. At a landscape scale, the groups are divided broadly between River Red Gum lined watercourses (Group 3) in the sandy northern bedload rivers and rivers dissecting the south-western stony plains and tablelands (Table 3-11), dominated by Coolabah–Gidgee woodlands. In the latter case, Coolabah (Group 2) dominates along higher-order streams and near the main channel and Gidgee (Group 4) dominates on lower-order streams and the outer floodplain.

At a patch-scale, the south-western stony country supports a mosaic of species groups (Figure 3-27), reflecting the complex geomorphology and hydrochemistry of the floodplains, with single channel, anastomosing or poorly-channelized areas and non-saline and saline reaches (Section 2.4). Non-saline, anastomosing (e.g. Mathieson to Neales Corner) and single-channel reaches (e.g. Algebuckina to Tardekarinna) support Lignum and Coolabah (occasionally River Red Gum) lining the deep waterholes (Figure 3-25), grading into sparse Coolabah-Gidgee woodlands over tussock grasslands on the poorly channelized downstream floodouts (Figure 3-27). Rich grasslands (Barley Mitchell Grass) or low chenopod shrublands (Bladder Saltbush) (Group 4) and occasionally Canegrass swamps (Group 1) form in depressions away from the channel. Within these depressions, small flood runners cross the floodplain, supporting *Acacia* spp. or Coolabah. Conversely, along the multi-thread or anastomosing channels of non-saline and saline floodplains, areas of high microrelief are formed through deposition of alluvial clays supporting Gidgee and Ruby Saltbush, or Nitre Bush in saline areas (Group 4). Low-lying areas support low chenopod shrublands or grasslands with occasional Coolabah.

The analysis did not associate salt-tolerant species with a specific group; rather these are included with the outer floodplain Group 4. This may be due to variation in microrelief and soil conditions, occurring at a smaller scale than the survey quadrats allow. Outer floodplain surveys indicate a combination of drought-tolerant species and less salt-tolerant species on rises and more salt-tolerant, flood-tolerant species in depressions (see Appendix D.4).

[12] *Ordinations show assemblage groups (1–4) based on scaled abundance data (Appendix A). Each number assigns a group to site based on dissimilarity (Bray-Curtis index), where nearby points are most similar. The righthand plot overlays indicator species, with higher values for the assigned group. For species abbreviations, see Table 6-10.*

Table 3-11: Permutational ANOVA analysis (Adonis) showing a significant effect of catchment (Finke, Macumba, Neales, Kati Thanda–Lake Eyre South) on species groups, independent of landform

Effect	df	SumsOfSqs	MeanSqs	F.Model	r^2	P
Catchment	3	3.7179	1.23931	4.5005	0.21173	0.001
Landform element	1	0.3489	0.34892	1.2671	0.01987	0.213
Residuals	49	13.4932	0.27537		0.7684	
Total	53	17.5601			1	

Figure 3-25: Lignum and Coolabah-lined waterhole typical of the WRLEB (photo: D Deane)

Few flood-regime indicators showed significant relationships with species groups (Table 3-12). The relationship with elevation indicates that landform setting (potentially a surrogate for stream order) has a strong influence, seen where Gidgee-dominated woodlands (Group 4) along lower-order rocky headwater streams (higher elevations) are replaced by Coolabah woodlands (Group 2) on higher-order floodplain channels (lower elevations) (Figure 3-26). The higher elevation of Group 3 River Red Gum woodlands does not indicate an elevated position in the landscape but the elevation of the northern rivers where the species mainly occurs. The other distance- or elevation-based measures, although drivers at a landscape scale, do not explain patch-scale variation. As an alternative, landscape GIS variables (Table 3-12) could be used to spatially model the distributions of environmental variables that support different vegetation types.

The Stony Plains Bioregion has an assemblage clearly influenced by distance to water (SAALNRMB 2010) and relief along a landscape-scale topographic gradient (Appendix E), but these relationships do not hold at the patch-scale (i.e. within a floodplain). This is because the survey methods do not capture data at a resolution that reflects the spatial arrangements of vegetation on microrelief. Subtle topographic variations are critical for other arid-zone plant communities (e.g. patch-interpatch dynamics in stony gilgai country: McIlwee et al. 2013).

Table 3-12: Results of envfit analysis

Landscape GIS variable	NMDS1	NMDS2	r^2	P
Distance to watercourse	-0.746	0.665	0.005	ns
Distance to waterhole	1.000	-0.023	0.036	ns
Elevation of watercourse	-0.373	0.928	0.331	<0.001
Elevation change to watercourse	-0.414	-0.910	0.038	ns

Patch scale variable	NMDS1	NMDS2	r^2	P
Site slope	0.994	0.109	0.025	ns
Strew size	0.687	0.727	0.015	ns
Strew cover	0.937	0.348	0.083	ns
Litter cover	-1.000	-0.012	0.053	ns
Percent clay	0.548	-0.837	0.187	0.031

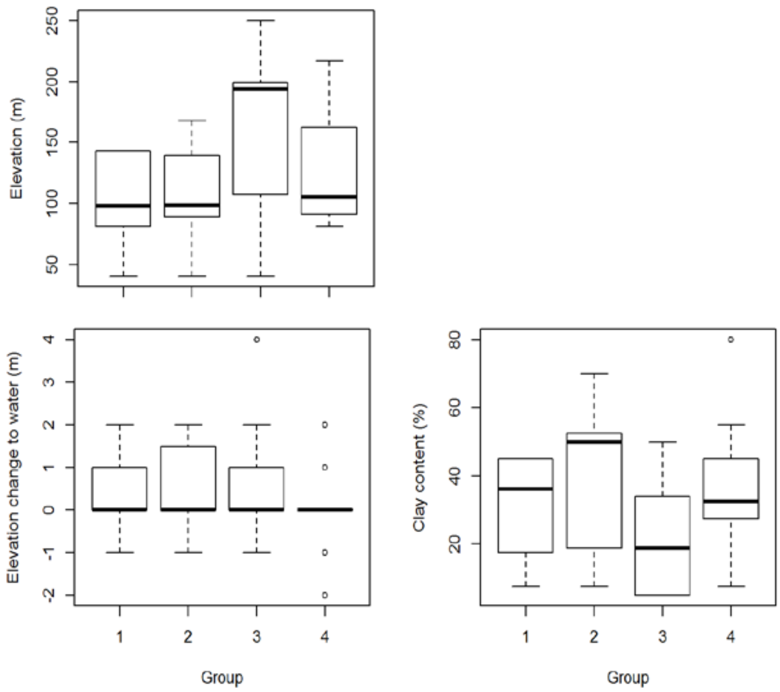


Figure 3-26: Variation in environmental variables for flood-dependent plant groups

Figure 3-27: Contrasting vegetation of the Macumba and Neales floodplains

Above, A 'standard floodplain' parallel zonation of River Red Gum–Coolabah–Gidgee (blue, yellow) and Gidgee-lined tributary (orange) of the Stevenson River (Macumba). *Below*, A 'non-standard floodplain' mosaic of Coolabah–Gidgee woodlands over perennial tussock grassland on the Neales at Stewarts Waterhole.

3.4.5.2 Modelling a 'non-standard floodplain'

The term 'non-standard floodplain' describes the mosaic of vegetation groups on floodplains of the central arid-zone rivers. The floodplains are formed by 'mega-floods' that re-set landforms over decades and centuries (Pickup 1991; Section 2.4), with associated erosion and deposition of sediments (Costelloe 2011; Wakelin-King 2011). These events destroy existing habitats and create new ones, initiating a succession of floodplain species. Cattle grazing and associated bank erosion are another major disturbance. Thus, a trajectory for the modern floodplain is difficult to determine, being shaped by major flows, sediment pathways and microrelief, supporting a habitat mosaic and a complex species assemblage. The species distributions also vary through expansion and contraction of suitable habitats, evidenced by Coolabah and Gidgee 'death zones' on the floodplain (e.g. Figure 1-2).

An approach to better understand how the microrelief positions of vegetation groups relate to flow regime was trialled using modelled stage height (Montazeri and Osti 2014) and floodplain elevation cross-sections. The results are preliminary, given the unreliability of the modelled Average Return Interval (ARI) for stage heights above cease-to-flow, but the method could prove useful. The following is a summary of the analysis (for details, see Appendix E).

Modelling suggests that the four focus reaches receive floods of different magnitudes (flow bands) and different return times (Appendix E). The shallow, multi-thread channels of the Arckaringa floodplain are inundated at a maximum modelled flood-level of 1.5 m above cease-to-flow, compared to 6-7 m for the deep single- and multi-channels of the other three reaches. Much of the variation in water levels is explained by the width: depth of floodplain channels (i.e. area of discharge), shown in the stage-discharge curves for the four sites. The wide (2. 4 km), shallow Arckaringa floodplains are unconstrained, so that floodwaters spread over a broad area under relatively low-flow conditions, whereas the deep, confined channel of Algebuckina Waterhole is likely to hold larger, less-frequent flows, with some water re-routed into parallel overflow tributaries. The Peake is another example of a confined channel, but is relatively shallow (4.7 m). Stewarts has characteristics of confined and unconfined channels, in the reach from Mathieson to Neales corner (Appendix C), and receives overbank flows at a higher frequency. The unconfined channels downstream of refuges (South Stewarts, Warrawaroona, South Cliff) probably receive floods at semi-regular intervals (Figure 6-15).

Cross-sections (Figure 6-15) indicate Gidgee-dominated 'hummocks' are on microrelief above the surrounding floodplain and would be inundated less frequently than the adjacent flood runners. The Arckaringa floodplain has many of these hummock areas, likely formed during floods that scoured and deposited alluvial, clay-rich sediments. This may mean that floods that inundate broad areas of floodplain for long periods discourage the survival and recruitment of Gidgee, as the species requires elevated microrelief. Conversely, the high-flow bands of Arckaringa Creek may represent unfavourable conditions for growth and establishment of Coolabah, a species requiring more frequent flooding (at least 1:5 years: Roberts and Marston 2011). There is sparse Coolabah woodland on the floodplain, with trees mainly along flood runners or incised channels that would receive more regular flows, suggesting that they rely on residual bank storage between flows (Costelloe et al. 2008).

The saline reaches of the Peake catchment support assemblages dominated by salt-tolerant species (Group 4). From limited data, these environments receive large floods exiting the confined channels of the Peake-Dennison Ranges. In general, low chenopod shrublands occupy lower microrelief (e.g. Arckaringa Creek), suggesting that these species tolerate longer periods of inundation than trees and tall shrubs (*Acacia* spp.) on elevated microrelief, or that they regenerate after flooding (cf. Capon 2005). Similarly, the floodplains of non-saline reaches support tussock grasslands at lower elevations, and potentially receive regular, high-magnitude floods. Examples are on the floodplains of the upper Neales (Stewarts, South Stewarts).

Grassland and shrubland assemblages in low-lying areas receive regular, shallow floods and deep inundation by infrequent, big floods. In broad, poorly-channelized floodplains downstream of the confined channels, there are diverse mixtures of chenopods, grasslands, swamps and Coolabah-Lignum lined flood runners. This is seen in the NDVI time series for floodouts at Stewarts, Algebuckina and Peake waterholes, showing green in response to flow (see Appendix C). The evidence for high flows in these areas includes tall Lignum (2 m) in channelized sections downstream of Algebuckina. In contrast, the floodplains of Algebuckina and Stewarts (and to a lesser degree Peake), with deep confined channels that overflow less often, support a sparse understorey of unpalatable species like Nitre Bush, due perhaps to grazing and/or the flow regime. Grazing may be intensified by the presence of persistent water at Algebuckina and Stewarts.

The role of regular flooding in maintaining high-value pasture (e.g. Bladder Saltbush, Mitchell Grass) is reflected on stony gilgai, where many species rely on depressions receiving runoff from the surrounding gibber pavement (McIlwee et al. 2013). The productivity of shrubs, grasses and Coolabah also may be reliant on maintenance of microrelief (or processes creating new microrelief), with run-ons transferring water and nutrients to smaller floodplain areas sustaining plant and animal populations and habitats (Ludwig et al. 1996). This is supported by NDVI 'hotspots' in response to flow, persisting through drought (Section 3.4.5). Future studies could ground-truth these hotspots to confirm their setting and investigate other potential water sources.

A preliminary sap-flow monitoring trial at various locations of the Neales–Peake catchment confirmed that Coolabahs at Algebuckina access shallow alluvial groundwater recharged from direct infiltration and streamflow (Ryu et al. 2014). At other sites, where trees were growing over highly saline (mid-reaches) or deep groundwater, they were highly responsive to streamflow (Ryu et al. 2014). This highlights the drought tolerance of the species and its capacity to use streamflow, bank storage or groundwater (Costelloe et al. 2008), as do River Red Gums along other rivers (Mensforth 1994). Estimations of the flooding requirements for Coolabah therefore require knowledge of surface-groundwater sources on a site-site basis. An alternative approach to predict vegetation composition could be to investigate relationships between soil properties, surface flow regimes, groundwater and vegetation.

The zonation of vegetation along elevation gradients may provide an indicator for changes in flow regime. Investigations are needed to develop more-refined models of inundation frequencies and population viability, particularly for long-lived perennial species. This could build on remote-sensing analysis, and be supplemented by data for tree and shrub water use.

3.4.5.3 Aquatic plants (Group Sp)

Aquatic plants were not recorded in the surveys, although this could reflect a bias towards non-aquatic habitats and the ephemeral nature of regional aquatic habitats. Perennial stands of aquatic plants do occur in spring-fed and bore-drain wetlands associated with GAB discharge. The spring complexes of the Neales–Peake catchment (Mt Dutton, Nilpinna, Freeling) are in elevated areas and infrequently connected to rivers. Satellite imagery shows that Freeling Springs were connected to the Peake River three times in 1999–2013, during catchment floods in 2000 and 2010–11 (Appendix C). Thus, transient water and infrequent connectivity limit the distribution of aquatic species, seen in the differentiation of base-cut survey plots of riverine environments and spring-fed wetlands based on assemblage structure and indicator species (Figure 3-28).

Bore-Drain Sedge and Common Reed occur in spring-fed wetlands, with Bore-Drain Sedge in permanently wet areas and Common Reed in drier areas (Table 3-10), reflecting their water requirements (Roberts and Marston 2011). Both are restricted to springs and bore-drain wetlands in this region, but Common Reed also occurs in other temperate riparian and floodplain ecosystems where flows are stable or regulated. Under more constant flows, these species are likely to become invasive and dominant, to the detriment of existing assemblages.

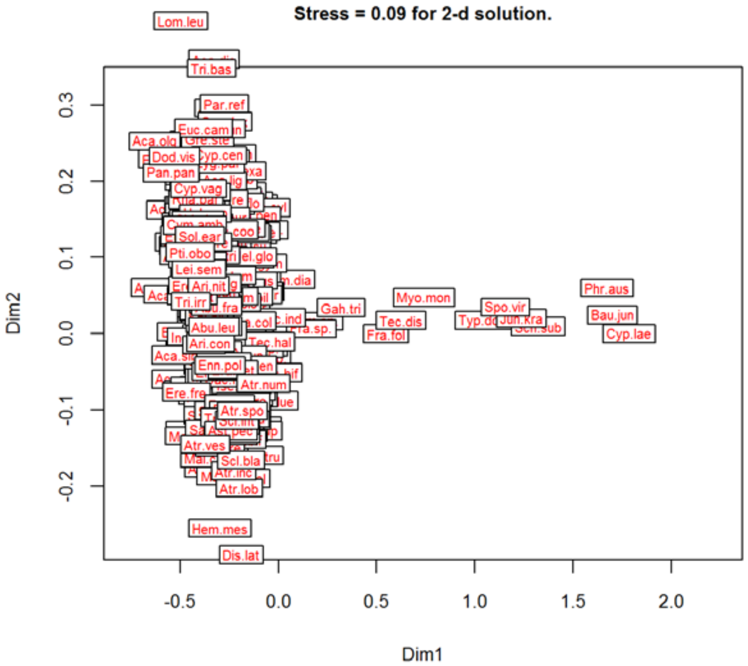


Figure 3-28: Ordination of plant species (*n* = 419), showing spring and non-spring groups. The visible species have higher indicator values (for abbreviations, see Table 6-10)

4. Potential impacts of mining

This section outlines a scenario-based, conceptual framework to guide assessments of the hydroecological impacts of mining in the WRLEB. The terminology of pressures and stressors aligns with the pressure-stressor-response model framework which has been incorporated in the Integrated Strategic Management Framework developed for the LEBRM project (McNeil & Wilson 2014).

4.1 Pressures

CSG and mining development in the WRLEB will affect the flow regimes supporting WDEs. The salient question is whether these pressures will operate within the tolerances of the ecosystems and their biota (cf. Bond et al. 2008).

Three scenarios may occur:

- Scenario 1: no diversions (e.g. impoundment of watercourses),
- Scenario 2: discharge of dewatering or co-produced water, and
- Scenario 3: full diversion using artificial channels or pipelines (i.e. diversion around mine pits or infrastructure).

These scenarios are related to four abiotic (A1 to A4) and three biotic (B1 to B3) pressures, described in Box 1. The pressures are largely concerned with surface waters as the interactions between surface water and groundwater (both shallow tertiary and deeper GAB) are poorly understood in the WRLEB. The presence of the relatively impervious Bulldog Shale (GAB aquitard unit) and upward hydraulic gradients from the underlying GAB indicate the GAB is largely disconnected from surface water systems in the WRLEB other than at select locations where GAB springs and zones of preferential discharge occur within and near rivers and floodplains. Understanding of surface water-groundwater interactions will be further advanced through another water knowledge project, the LEB Springs Assessment (DEWNR 2015).

Box 1. Pressures from CSG and coal mining scenarios in a conceptual model for WRLEB rivers
Pressure PA1: Drought
Impoundments associated with mining may reduce flows and impose drought-like conditions on watercourses (Davis et al. 2006), and diversions and point abstractions would have similar effects. Although drought is a natural pressure in the arid zone, it has adverse effects on ecosystems, especially when prolonged (e.g. Magalhaes et al. 2007; Bond et al. 2008; Benejam et al. 2010). For example, diversions may affect riparian vegetation through lack of bank recharge, and changes in water level may cause bank collapse and sediment mobilization (PA3). The effects may be countered in subsequent wet seasons if the system's resilience (its capacity to recover) is preserved.
Pressure PA2: Constancy
Constant discharge into watercourses (e.g. co-produced water, or dewatering) reduces natural flow variability (Bunn et al. 2003; Sheldon et al. 2010; Arthington and Balcombe 2011). As many arid-zone species and processes are adapted to variable, intermittent flows (SB1), constancy is a pressure.
Pressure PA3: Erosion and sediment loads
Sediment mobilization causes silting in waterholes and threatens species reliant on the few deep refuges that sustain populations through drought (McNeil et al. 2008; Wakelin-King 2011). Sediment transport issues are more common in the northern bedload rivers that lack refuge waterholes, but impoundments and diversions eliminate the flow pulses that scour waterholes, and they may also decrease bank storage and affect the long-term vigour of riparian trees, leading to loss of bank integrity and collapse. Constancy is a low-energy regime, but it may affect bank integrity by undercutting.

Pressure PA4: Salt balance
Reduced flows (Scenarios PA1–PA2) and/or constant discharge (PA2–PA3) may cause salt to accumulate rather than be flushed downstream, causing habitats to become saline (Scenario 1), or it may redistribute salts along mid- and downstream reaches (Scenarios 2–3). In addition, discharge waters may differ in ionic composition or contain chemicals, including heavy metals, that impact on diversity and biotic integrity (Davis et al. 2010).
Pressure PB1: Competitive advantages for species
Variable flows allow partitioning of resources in space and time and promote coexistence of species. Lost connectivity with ephemeral habitats will decrease breeding and recruitment opportunities, and may cause a decline of mobile 'periodic' species (Spangled Grunter, Bony Herring) and 'opportunists' (Desert Rainbowfish, Golden Perch). Constant (regulated) flows favour the spread of Eastern Gambusia and Common Reed into discharge sites and watercourses (Scenarios 2–3). By contributing to altered salt balance, hydrologic changes are likely to promote 'specialist' species in Polo Club refuges to the detriment of more diverse assemblages in Ark refuges (e.g. Algebuckina Waterhole).
Pressure PB2: Floodplain and riparian habitat integrity
Habitat integrity in riparian corridors and floodplains is part of a healthy river ecosystem (Pusey and Arthington 2003). Mining and CSG development may affect habitat integrity directly (physical disturbance) or indirectly (lack of flooding, isolation by infrastructure), and threaten vital food resources and breeding habitats (Balcombe et al. 2007).
Pressure PB3: Introduced grazers (cattle)
Mining and CSG development may affect patterns of cattle grazing by altering the distribution of water and WDEs in the landscape. This would affect floodplain and riparian vegetation as well as geomorphic processes (cf. Wakelin-King 2011).

4.2 Stressors

Following the review of concepts (Section 1.4), two abiotic (A1, A2) and two biotic stressors (B1, B2) are identified in the conceptual framework shown in Box 2.

Box 2. Stressors in a conceptual model for WRLEB rivers
Stressor SA1: Catchment floods
Catchment-scale floods reset moisture and energy reserves. Floods drive productivity and create temporary resource-rich habitats and opportunities for growth and recruitment of flood-dependent species. Bankfull flows also recharge bank storage and thereby maintain bank integrity.
Stressor SA2: Hydraulic connectivity between springs and rivers
Hydraulic connectivity between springs (or bore drains) and rivers enables salt transport between middle and downstream reaches (Costelloe et al. 2005b). The accumulation and movement of surface salts create naturally-saline Polo Club refuges, with distinctive aquatic assemblages and a mosaic of chenopod shrublands and grasslands providing diverse habitats and fodder for grazing stock.
Stressor SB1: Flow variability
Flow-no flow periods support the recruitment and dispersal of opportunistic, periodic and specialist aquatic species (Arthington and Balcombe 2011). They also promote moderate recruitment for opportunists like Desert Rainbowfish and Golden Perch, and limit the spread of Eastern Gambusia, which is reliant on stable, spring-fed pools. Periodic connections during sub-catchment or catchment flows create new habitats (Disco refuges, temporary wetlands) for mobile species like Spangled Grunter and Bony Herring. Flow variability, with PA1 and PA2, redistributes salts, creating Polo Club refuges that support specialists like Lake Eyre Hardyhead. Flow variability is significant for recruitment and dispersal of many species, including fish (Costelloe et al. 2003, 2007).

Stressor SB2: Flow regimes supporting patch-interpatch dynamics
Frequently inundated areas occur downstream of waterholes (large-scale run-on patches *sensu* Ludwig et al. 1996) or near channels and waterholes. These areas support River Red Gum or Coolabah grassy woodlands, whereas less often flooded, outer areas support drought- or salt-tolerant species (e.g. Gidgee). Woodlands provide fodder for stock in wet years; outer floodplain species provide fodder in drought years. Over-grazing may affect recruitment windows for terrestrial species through interactions with flooding (Reid et al. 2011).

4.3 Components (assets and receptors)

Habitats supporting fish and other aquatic biota of the western rivers can be conceptualized as 'Ark', 'Disco' and 'Polo Club' refuge types (Robson et al. 2008), each with a distinctive hydrological regime, core assemblage and indicator species. Ark refuges are rare, as most waterbodies are shallow (< 2 m). Disco refuges are common in the upper reaches, some providing 'stepping stones' for biota under high-flow conditions and others persisting for several years following floods. Polo Club refuges are in the lower reaches, downstream of saline reaches (e.g. mid-Peake, Neales); they generally are saline (> 20 ppt) and harbour a small complement of salt-tolerant species.

Springs and bore-drain wetlands also provide refuges for aquatic biota, but only a small proportion of riverine species utilise them due to their shallowness (< 0.3 m) and peculiar water chemistry. They are habitats, however, for Eastern Gambusia (McNeil et al. 2011). Dams also are likely to support aquatic biota, but they have poor water quality and lack diverse habitats (McNeil et al. 2011). Floodplains provide mixed pastures of drought- and salt-tolerant species as habitats and fodder for water-dependent terrestrial biota, and food and breeding grounds for aquatic biota.

Reference conditions for the components of water-dependent ecosystems are likely to prevail only during a baseline no-flow phase (Stage I, Section 3.4.2). Simultaneous or sequential flows (local, sub-catchment, catchment scale) connect these habitats, changing assemblages in habitats that otherwise support distinctive species (e.g. Ark, Disco, Polo Club refuges). Pulsed flows promote ecosystem stability, forming temporary 'highways' used by species to colonise new or once-occupied habitats (thereby countering local extinctions) and redistributing water and nutrients to productive patches. Inter-habitat connections are vital to sustain biota, as in the eastern basin, but their role in the western rivers is emphasized because the greater ephemerality of waterways drives wider population fluctuations, making species more vulnerable to local extinctions (McNeil et al. 2008). In addition, longer return intervals mean that high-flow stages (catchment floods, Stage IIC) are critical for long-term recharge of deep-water stores and in support of patch-interpatch dynamics in flood-prone terrestrial habitats.

Key model components are shown in Box 3.

Box 3. Components of a conceptual model for WRLEB rivers
Component H1: Ark, Disco and Polo Club refuge habitats Component H2: Spring (and bore-drain wetland) habitats Component H3: Non-saline and saline floodplain habitats Component S1: Water-dependent aquatic biota (represented by fish and vegetation indicator species) Component S2: Water-dependent terrestrial vegetation (tree, shrub and grass indicator species)

4.4 Response models

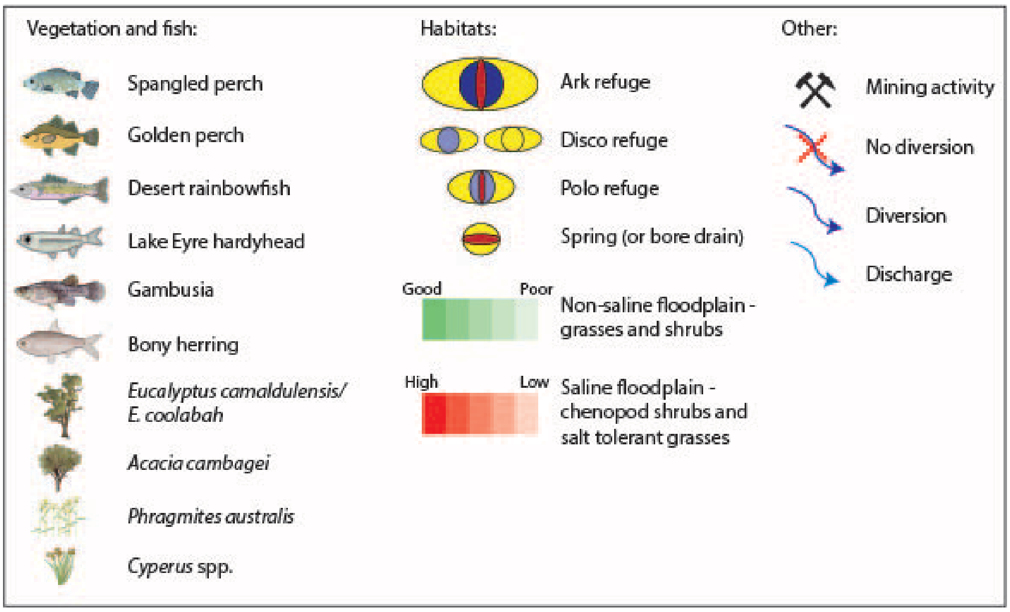
From the foregoing (Section 3), conceptual models are presented to demonstrate the potential impacts on pressures and stressors and the responses of components of WDEs to natural and altered flow regimes under the three mining scenarios (Figures 4-1 to 4-4). These are preliminary and qualitative, and need refinement particularly through hydroecological modelling as well as investigations into surface water – groundwater interactions. They have been used to identify indicators and response types in combination with the review of concepts (Section 1.4), and could be used in monitoring and modelling the impacts of mining and CSG development (Section 5.1). They could later be extended through quantitative modelling to inform impact assessments, and in development of Bayesian models.

The response components are confined to the two biotic groups studied in Section 3, and future efforts could also extend these models to incorporate other biotic groups (particularly invertebrates) and abiotic processes (particularly organic matter processing and nutrient availability). However, as discussed earlier, new data would be required to model their responses. Another LEBRM project (Imgraben and McNeil 2014) developed generic hydro-ecological and pressure-stressor-response models for different LEB aquatic ecosystem types which include a broader range of biotic groups and stressors and illustrate the relationships between pressures, stressors, components and processes. These models (Imgraben and McNeil 2014) provide a framework to integrate other components.

Mining and CSG developments may impact on river ecosystems through pressures (Box 1) driven by changes in catchment-scale floods (Scenarios 1–2) and water and salt balance (Scenarios 2–3). Changes in community composition are likely through loss of hydroecological functions and ephemeral wetlands that support recruitment and dispersal of mobile species and riparian species and affect the competitive advantages of alien and native species. Grazing pressures are mediated through changes in floodplain productivity, carrying capacity and bank water storage and integrity.

The conceptual models suggest that changes would occur in the hydroecological integrity of rivers under each mining scenario. Scenario 1 (Figure 4-2) has the most severe impacts, with impoundments causing drought-like effects, leading to the decline of species *via* loss of connectivity (SA2), and with loss of floods (SA1) causing a decline in flood-dependent riparian species, compromising habitat integrity (PB2). If this were to occur in the Peake sub-catchment (where most coal deposits are located), it would change the salt balance (PA4), leaving a sparse fish assemblage in a few isolated, Polo Club refuges. Scenario 3 (Figure 4-4) has the least severe impacts, with some increase in floodplain productivity but promotion of undesirable species (e.g. Eastern Gambusia, Common Reed) (PB1). Scenario 2 (Figure 4-3) has intermediate but still severe impacts, with hydrologic constancy (PA2) favouring alien species (Eastern Gambusia) in spring-fed pools (PB1).

Legend for model scenarios presented below



(Note: for 'disco refuge' blue centre relates to connectivity)

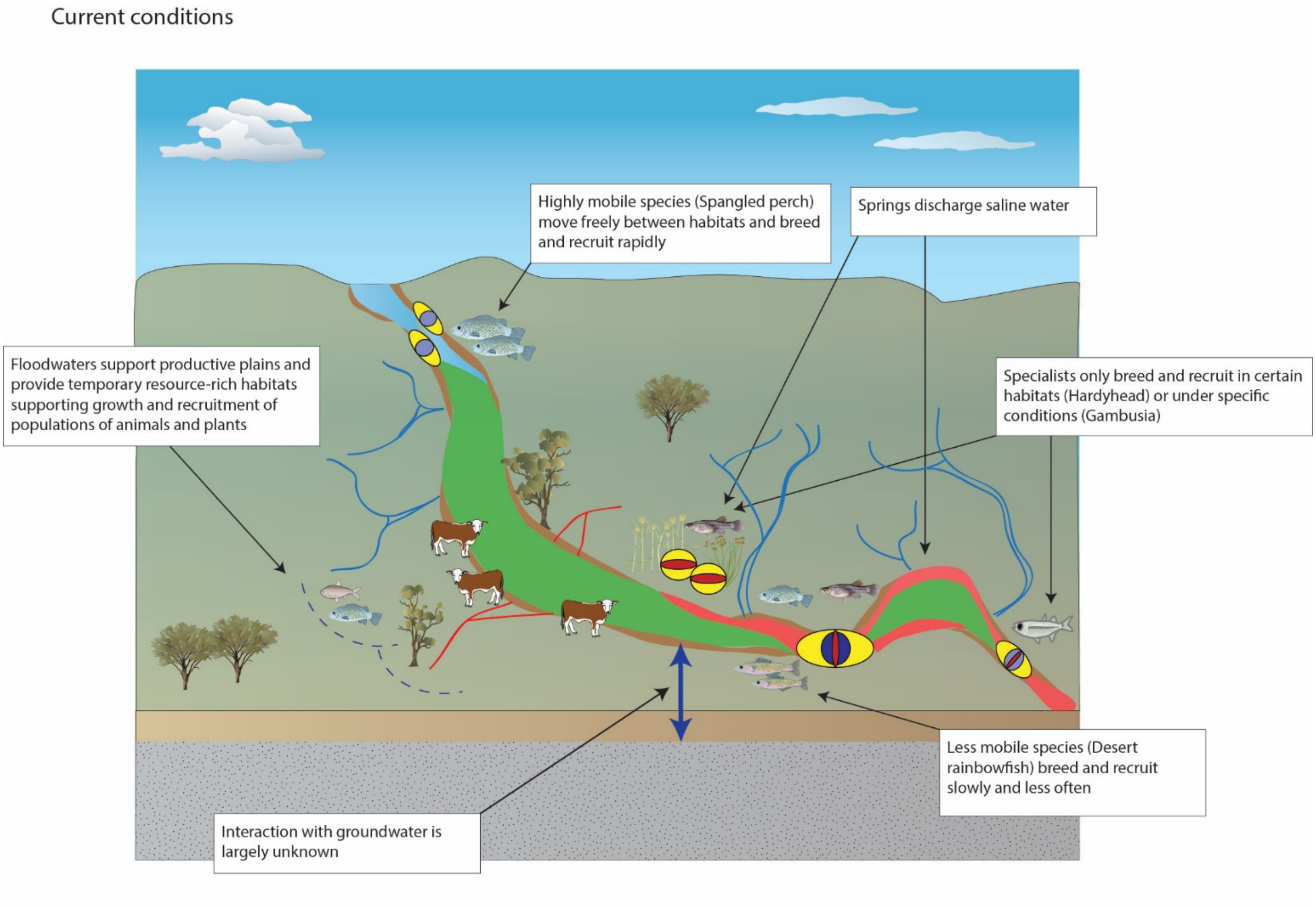


Figure 4-1: Hydroecological components, stressors and pressures under present conditions

Note that the floodplains support increased populations of stock and other grazing animals during dry periods, when water supplies are scarce and plant growth is reduced (Section 2.1).

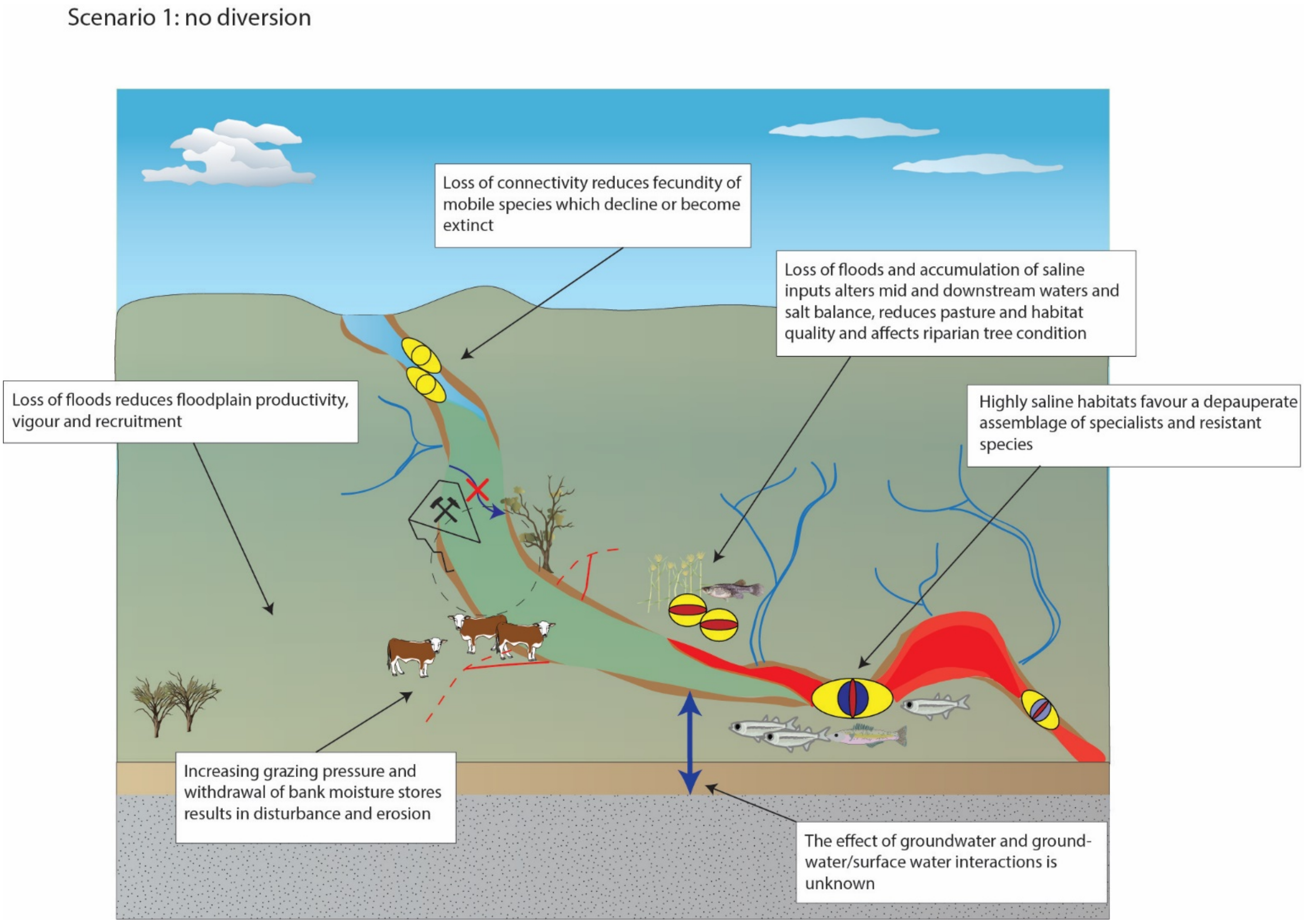


Figure 4-2: Hydroecological components, stressors and pressures under Scenario 1: no diversion (e.g. impoundment of flow without diversion around mine site). This model applies to catchments with saline reaches (e.g. Peake)

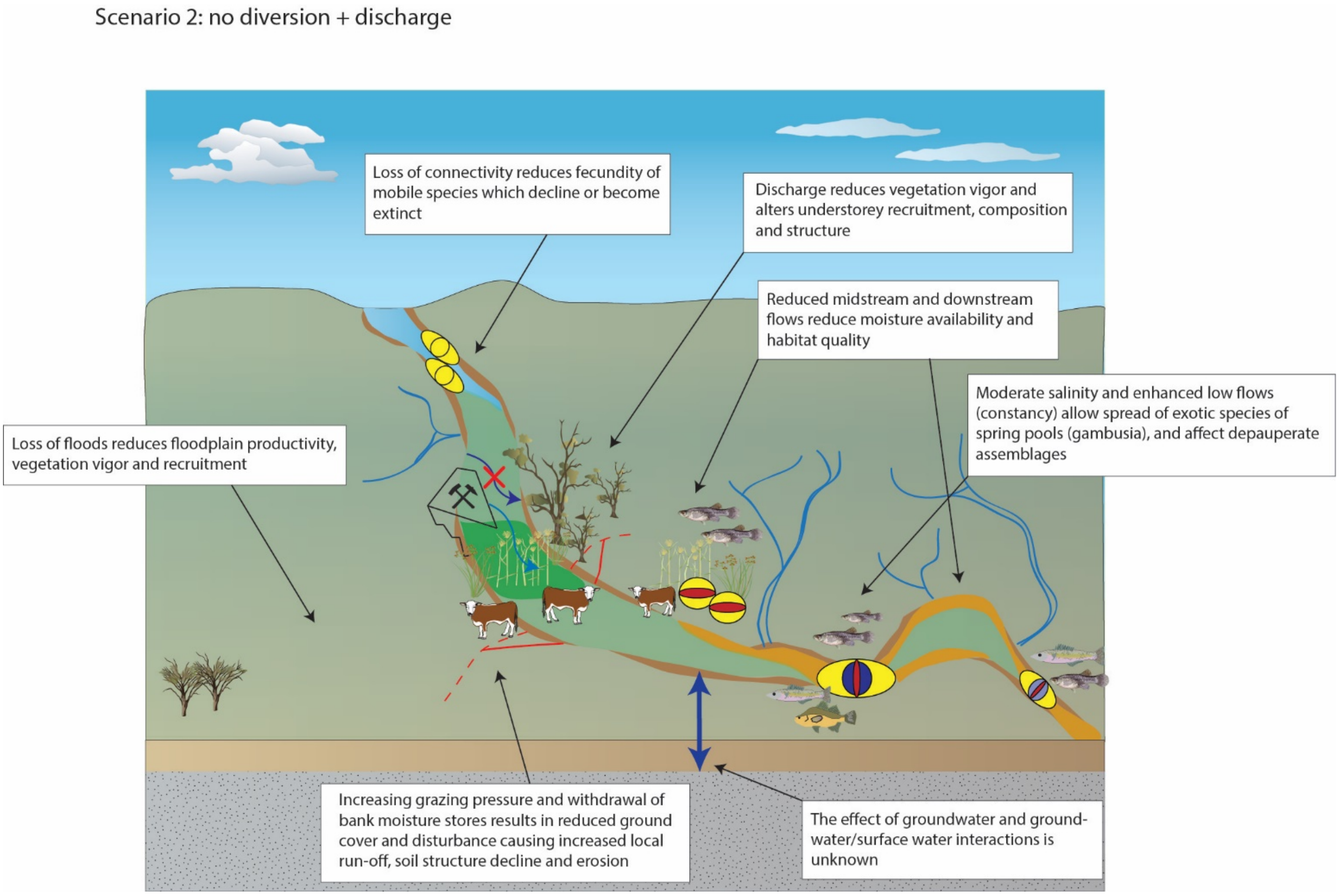


Figure 4-3: Hydroecological components, stressors and pressures under Scenario 2: discharge (e.g. co-produced water) and no diversion (e.g. upstream impoundments)

Scenario 3: Full diversion

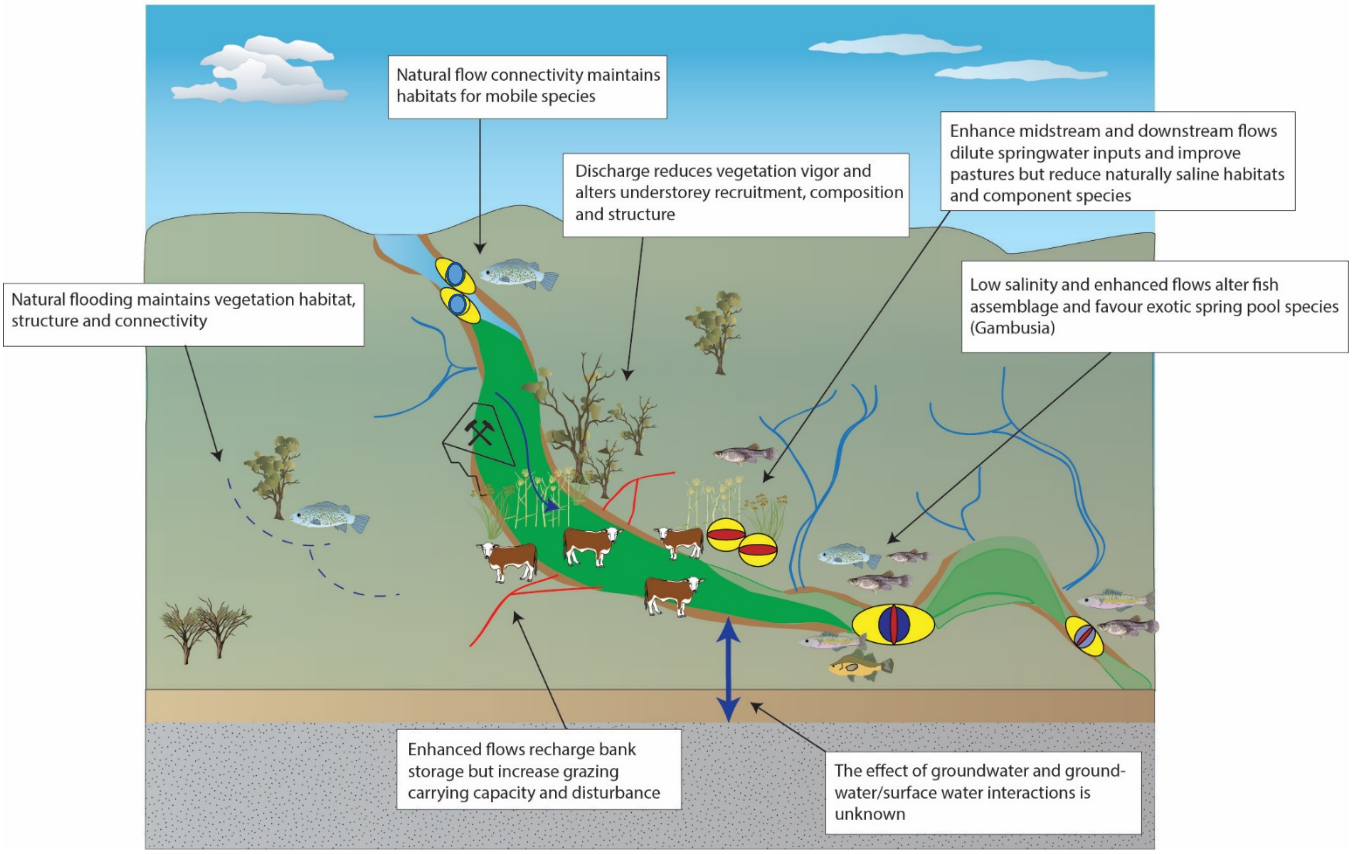


Figure 4-4: Hydroecological components, stressors and pressures under Scenario 3: full diversion and discharge

5. Conclusions and recommendations

5.1 Indicators and response types

The foregoing analyses are an aid to understanding the dynamics of WRLEB ecosystems by incorporating hydroecological components, stressors and pressures. The Bioregional Assessment Programme, for the Pedirka and Arckaringa subregions, will make an assessment of potential impacts on these types of ecosystems from possible coal resource development. The principal effects are likely to be related to hydrological changes, and evident in the responses of certain species that indicate the integrity of aquatic and terrestrial ecosystems.

This study considers potential indicators and responses identified by modelling hydrological, fish and vegetation data. Quantitative indicators and response types are identified in Table 5-1, with likelihood ratings for responses under three mining scenarios (Section 4). The current state of the indicators is a baseline for assessments. Steps toward predicting potential impacts could be made using spatially-explicit or time-series models under different scenarios (e.g. Appendix D.5). Ideally, the potential indicators and response types could be identified more precisely and linked in the form of testable (falsifiable) hypotheses, with appropriate bounds in space and time. Once working hypotheses are identified, all available data and supporting literature should be reviewed before proceeding to an empirical test.

The indicators (Table 5-1) are proposed as a starting point for risk assessment and design of monitoring programs, but they need refinement to define baseline status, thresholds and timeframes, depending on the circumstances (e.g. nature of the impact, the indicator and likely response). Feedback loops will occur between responses and should also be considered. Some indicators are likely to prove more sensitive than others and may have special significance for early warning, compliance or diagnostic monitoring. Some may show a time-lagged response, making it difficult to determine when their responses are outside the normal range of variability (Sheldon et al. 2005, 2012). In these cases, abiotic indicators may prove useful.

The hydrological regimes of water-dependent habitats could be used to identify attributes for assessment of risk at a landscape scale (Miles and Miles 2014). Catchment-scale analysis supports conceptual models of baseline hydrological regimes at the scales of catchment floods, sub-catchment connecting flows and local recharge flows (Costelloe et al. 2007). Groundwater contributions are little-known (Miles and Miles 2014) and need to be investigated in order to understand the vulnerability of aquatic ecosystems to changes in groundwater conditions.

This study is focussed on fish and vegetation, the two groups for which there are useful data, but these are not intended as proxies for the responses of the entire system. The indicators proposed below (Table 5-1) are not comprehensive, and do not imply that impacts on other biotic groups should not be considered.

Pressures

According to the conceptual models, CSG and coal mining development potentially will impact on WDEs in four areas:

1. Increased low flow permanency,
2. Reduced flow variability,
3. Reduced flood frequency and magnitude, and
4. Salinity and river salt balance.

Other disturbances are not considered explicitly here, but could compound the effects of altered flow regimes. They could include changes in patterns of grazing and the climate.

Response types

The pressures of CSG and coal mining development are assigned response types related to vulnerabilities in the hydro-ecological integrity of the WDEs. These are in four groups (Section 3.4): (1) altered flow facets, (2) loss of habitat integrity, (3) changes in the distribution and composition of fish and vegetation assemblages and (4) changes in breeding and recruitment dynamics.

Indicators

Each response type is assigned one or more indicators representing the natural flow regime, fish and vegetation and habitat integrity. A similar approach has been used in water-allocation planning in South Australia (e.g. Green et al. 2014), maximizing detection of changes in hydroecological integrity that may not be understood at a mechanistic level. The types of analyses applied here could be improved by research, modelling and monitoring, and the suggested indicators and measures should be seen as a 'first pass' for managers considering how best to represent risks to hydroecological integrity in water planning.

Measures

Quantification of detection probabilities under various hydrological conditions (e.g. Clemman et al. 2013) is needed to estimate detection frequencies for fish indicator species (Section 3.4.4). Detection probability modelling may refine abundance or time-series measures for identifying mining-related pressures, and could complement the use of multivariate measures (e.g. the dissimilarity of any given site.visit from a group centroid, or a 'direction of species indicator' in Table 5-1). Uncertainties remain over the timeframes needed to detect significant changes and the appropriate spatial scales for measurements. Hydrological measures could be refined by modelling (e.g. time series of inundation: Appendix E). With these refinements, hydrological scenario modelling could identify thresholds for monitoring and management trials.

Likelihoods

The likelihood of a measure exceeding a threshold is a guide to track changes in hydroecological integrity. Likelihood ratings under the mining scenarios suggest that different thresholds and pressures arise from alternate management options—for example, flow constancy is a concern for discharge scenarios and declines in flooding and changes in salinity are important for impoundment scenarios. It is interesting that the discharge scenario registers a greater alteration than the two other scenarios, in accord with the dominant role of flow variability in maintaining hydroecological integrity (cf. Walker et al. 1995). The likelihood ratings need to be refined; this is a relatively straightforward process best applied through quantitative scenario modelling in eSource (D Penney and M Montazeri, DEWNR, unpublished data) or adaptive management trials. Trials could begin by monitoring small-scale developments (e.g. dams, road bunds) with reference to an experimental design, leading to statistical analyses of sensitivities and thresholds for indicators and measures.

Table 5-1: CSG and mining: potential vegetation, fish and hydrological responses, indicators and measures[13]

CSG/mining–related pressure	Response type	Indicator	Measure (thresholds)	Likelihood rating under scenarios		
				No diversion (1)	Discharge (2)	Full diversion (3)
Increased low flow	Increased flow constancy	Zero flow days (zqd)	Shape of exceedance probability curves	Moderate	High	Moderate
	Increased distribution and abundance in non-Ark waterholes (Disco refuges)	Eastern Gambusia	Change in detection frequency and direction of species indicator	Low	High	Low
		Desert Rainbowfish		Low	Moderate	Moderate
	Populations established in riverine ecosystems (esp. deep waterholes)	Common Reed	Presence in riverine habitat (>2% cover)	Low	Moderate	Moderate
	Populations established in riverine ecosystems (esp. shallow waterholes and in-channel)	Bore-Drain Sedge		Low	Moderate	Moderate
Decline in flow variability	Decreased patch variability	Normalized Difference Vegetation Index (NDVI)	Variation in NDVI over floodplain reaches	High	High	Moderate
	Decreased flow intermittency	Flow event frequency (qve.90, qve.730)	Shape of exceedance probability curves	High	High	High
		Flow duration (qd.90)		High	High	High
	Altered fish diversity	Diversity of fish assemblage groups	Diversity of groups (- 2 SD)[14]	High	High	Moderate
	Reduced breeding and recruitment	Spangled Grunter, Bony Herring	Reduced peak frequency above / below average spawning size	Moderate	Moderate	Low
	Decline in abundance/distribution (in Disco refuges)		Change in detection frequency and direction of species indicator	Moderate	Moderate	Low
	Decline in abundance/distribution (in Polo Club refuges)	Lake Eyre Hardyhead		Low	Moderate	High

[13] See Section 3 for further explanation of indicators and thresholds

[14] Decline of >2 SD (see Table 3-6)

CSG/mining–related pressure	Response type	Indicator	Measure (thresholds)	Likelihood rating under scenarios		
				No diversion (1)	Discharge (2)	Full diversion (3)
Decline in flood frequency and magnitude	Decreased catchment variability	NDVI	Variation in NDVI over catchment	High	High	Low
	Decreased flow magnitude	Flow magnitude and duration (qv.90, qd.90)	Shape of exceedance probability curves	High	Moderate	Low
	Loss of habitat integrity	Waterhole cease-to-flow depth (sediment infill)	Change in cease to flow depth (>1 m)	Moderate	High	Low
		Reduced riparian tree vigour	Change in diurnal sap flow, change in vigour/mortality rates of species	High	Moderate	Low
	Shifts in distribution of vegetation groups	Shifts in distribution of vegetation groups (esp. 2, 4) along micro-elevation gradients	Change in cover and direction of indicator species	High	High	Low
Increased salinity	Increased salinity in Ark and Disco refuges and floodplains	Water salinity, soil conductivity	Salinity in consecutive years (water >30 ppt, soil >200 mS/cm)	Moderate	Moderate	Low
	Loss of habitat integrity	Reduction in riparian tree vigour and pasture (loss of Mitchell grasses, Bladder Saltbush)	Change in diurnal sap flow[15], change in cover/vigour of species	Moderate	Moderate	Low
	Distribution of salt-tolerant species	Salt-tolerant chenopods (Bindyi, Ruby Saltbush, Nitre Bush)	Change in cover of species	High	Moderate	Moderate
		Lake Eyre Hardyhead, Desert Goby	Change in detection frequency of indicator species	High	Moderate	Moderate

[15] See Appendix 1: Ryu et al. 2014

CSG/mining–related pressure	Response type	Indicator	Measure (thresholds)	Likelihood rating under scenarios		
				No diversion (1)	Discharge (2)	Full diversion (3)
Changed salinity regime (increase in Disco, Ark refuges/decrease in Polo Club refuges	Changes in baseline salinity in relation to flow stage	Water salinity-flow[16]	Predicted deviation (reach-scale salinity–flow model)	High	Moderate	Moderate
	Increased distribution into Disco and Ark refuges	Lake Eyre Hardyhead, Desert Goby	Change in detection frequency and direction of indicator species	High	Moderate	Low
	Decline in abundance/distribution in Polo Club refuges		Diversity of Group 4 species (- 2 SD)	High	High	Moderate

[16] See Section 3.4.3: River salt balance

5.2 Recommendations

This study has identified a number of key knowledge gaps and next steps that need to be addressed to reduce the uncertainty in managing risks to WDEs from CSG and mining-related activities in the western rivers. The following recommendations are made, and are grouped by priority for the purposes of strengthening risk assessment and management decisions:

- Very high (foundational activities without which risk assessment and management decisions cannot be undertaken):
    - Develop a practical framework for assessing risks to surface WDEs from flow-regime changes, including those arising from coal resource development. This should integrate and build on the results of all the LEB water knowledge projects (DEWNR 2015), the state-wide vulnerability assessment project (Berens et al 2014) and DEWNR's risk assessment framework for water planning and management (DEWNR 2012). In particular, it should be specific for the region, related to aquatic ecosystem types in the region and capable of assessing risks from specific developments. Such a framework is required for the entire LEB for the Bioregional Assessment Programme as well as for other purposes.
    - Review potential indicators, refine measures and identify thresholds to a standard appropriate for monitoring and detection of impacts.
- High (activities which would substantially advance understanding of the vulnerability of aquatic ecosystems and/or improve the robustness of risk assessment):
    - Extend the analyses for fish and vegetation to other biotic groups, including algae, invertebrates (micro- and macro-invertebrates), waterbirds, amphibians, reptiles and mammals, and obtain data also for microbial processes.
    - Investigate groundwater resources and surface water-groundwater interactions (Section 1.4). Research by JF Costelloe (e.g. Costelloe et al. 2005b; 2008; Costelloe 2011; Costelloe and Russell 2014) has identified groundwater interactions in waterholes in the mid-Peake and Neales catchments, but there is virtually no information elsewhere in the WRLEB. The Atlas of Groundwater Dependent Ecosystems (BOM 2014) identifies many WDEs with moderate to high potential for surface or subsurface groundwater dependence in the Neales catchment, and some in the Macumba (the Atlas does not cover the Northern Territory).
    - Document refuges and refuge types (Ark, Disco, Polo Club) for the Macumba catchment. Fish surveys undertaken for this study and LEBRA show that at least one Ark refuge is likely to exist in the Macumba catchment, but its location is yet to be determined.
    - Refine hydrological models to determine the extent and frequency of inundation associated with plant assemblages (Appendix E). The approach trialled here could be used to model the impacts of changes in flood regime on floodplain vegetation, but is limited by uncertainties in modelled stage heights above cease-to-flow levels (cf. Montazeri and Osti 2014). A priority should be to improve the accuracy of modelled stage heights.
- Moderate (activities that would substantially improve the accuracy hydroecological analysis and modelling of the WRLEB):
    - Undertake floodplain-extent mapping for the WRLEB (cf. Miles and Miles 2014). This project has made progress through remote sensing, but more work is needed in regard to maximum extent and flood ARIs. Recent mapping of open-water detection frequency by Geoscience Australia may under-estimate inundation extent and frequency, owing to short-flow durations and the small size of many waterbodies in the WRLEB.
    - Extend sap-flow monitoring, initiated in another LEB Water Knowledge project (Appendix A, Ryu et al. 2014). With analyses of xylem water and groundwater, this could be insightful in regard to water use by floodplain trees and their vulnerability to flow-regime changes. This form of monitoring should be expanded to refine indicators (Table 5.1) and assess risks from CSG and mining scenarios.
    - Investigate the detectability and abundance of species in relation to different flow stages.
    - Apply knowledge from other catchments with caution, as western catchments have distinctive assemblages. In the absence of regional data, knowledge from the better-studied eastern LEB catchments (and even the adjacent Murray-Darling Basin) may be applied to the WRLEB, but within limits. The analysis of fish data showed that there are different assemblages in eastern v. western v. Finke catchments (Section 3.4.4.1), and potentially different responses by some species to flow stages in eastern v. western catchments.

5.3 Conclusions

Developments that alter the natural flow regime of the basin's rivers inevitably will have consequences for WDEs (Kingsford 2000; Bunn et al. 2006; Arthington and Balcombe 2011; Costelloe and Russell 2014). This study has contributed towards development of a conceptual and quantitative framework which, with other LEB Water Knowledge projects (DEWNR 2014), should reduce the uncertainty in assessing risks from water regime changes that may arise from CSG and coal mining. The next step is to develop quantitative risk planning and evaluation methods. The conceptual models and hydro-ecological indicators and responses identified here may be used to inform asset-receptor models (Section 4.4), asset attribution and identification of management triggers, as part of strategic adaptive management (Kingsford and Biggs 2012). Ultimately, an evidence-based risk assessment framework, such as developed for GAB springs (Green et al. 2013), will provide the most transparent platform to address the regulatory needs and uncertainties faced by decision-makers and resource managers.

To minimise the potential impacts of mining and resource developments, it will be necessary to balance the prospective economic and social benefits with the consequences for the environment (Kingsford 2014). There are decision support tools to aid managers in merging hydro-ecological, social and economic models (Matthies et al. 2007), and plans that balance the objectives can guide monitoring of strategic indicators used to progressively refine management (Kingsford and Biggs 2012). A strategic adaptive management approach will also enable new information to be incorporated as it becomes available, including from other projects being undertaken concurrently to this. Risk assessments and adaptive management require close collaboration between researchers and scientists, managers and policy makers. Indeed, risks are intensified where the principles of adaptive management are not followed (Walker et al. 1995):

> *Monitoring, research and management are a triad: the goal of monitoring is to identify pattern, research is to understand process, and management uses both kinds of information to balance supply and demand in the long term. As we are constrained by the paucity of historical data, future ecology will be so limited if we fail now to establish ongoing programmes to monitor environmental change. The responsibility for implementation rests with managers and governments, but the tasks of design and review fall to researchers.*

Current monitoring of the LEB is undertaken to analyse trends in key indicators (fish, water quality, hydrology), but the data presently available are insufficient for predictive empirical models. One challenge for ecologists is to provide flow rules describing the dynamics of these highly variable systems (Arthington et al. 2006). Another is that fish and vegetation alone are unlikely to reflect the full spectrum of changes potentially associated with CSG and mining impacts, and data for other biotic groups (e.g. algae, invertebrates, waterbirds) are needed to provide robust measures of patch-scale condition. Improvements could be made in the design of monitoring to encompass nested scales in the dynamics of desert biota (Boys and Thoms 2006; Lowe et al. 2006), and in understanding the role of detection probabilities in monitoring for cryptic species (e.g. MacKenzie 2005). Ultimately, monitoring data should be rigorous and used routinely in testing hypotheses and advancing conceptual and quantitative models. Otherwise, changes may go undetected and opportunities for wise decisions could be missed.

6. Appendices

A. Data sources, custodians and programs

Table 6-1: Fish monitoring visits and programs

Trip	Date of visit		Monitoring Program
	Start	*End*	
1	30/03/00	02/04/00	ARIDFLO[1]
2	03/08/00	10/08/00	
3	26/11/00	29/11/00	
4	30/03/01	04/04/01	
5	31/10/01	08/11/01	
6	28/03/02	31/03/02	
7	15/02/03	28/02/03	
8	07/12/07	12/12/07	LEBRA[2]
9	29/04/08	02/05/08	
10	02/12/08	04/12/08	
11	30/05/09	01/06/09	
12	12/11/09	15/12/09	SAAL Critical Refugia[3]
13	10/04/10	18/04/10	
14	14/12/10	14/12/10	
15	13/04/11	04/06/11	LEBRA[4]
16	27/11/11	01/12/11	
17	24/03/12	15/05/12	
18	30/09/12	03/10/12	
19	13/04/13	09/05/13	LEBRA/LEBRM[5]
20	28/10/13	28/10/13	

* Results of monitoring programs: [1]Costelloe et al. 2007, [2] McNeil et al. 2008; McNeil and Schmarr 2009, [3] McNeil et al. 2011), [4](Cockayne et al. 2012), [5]This study

Table 6-2: Fish monitoring data for this study (1=surveyed, blank=not surveyed)

Catchment	Site	abbr.*				Visit												
		C	*S*	*Wtl*	*Wh*	*8*	*9*	*10*	*11*	*12*	*13*	*14*	*15*	*16*	*17*	*18*	*19*	*20*
Cooper	Cullyamurra	Coo	cul	wh	A		1		1			1	1	1	1	1	1	1
	Cuttapirra		cutt		D							1	1	1	1			
	Lake Hope		hop		D							1	1	1	1	1	1	
Diamantina	Pandi Pandi	Dia	pand	wh	A				1				1		1		1	1
	Wadlarkininna		wad		D										1			
	Cowarie Crossing		cow		P								1			1		1
Finke	Pioneer Creek	Fin	pio	wh	A								1		1			
	Running Water		run		A								1		1		1	
	Main Camp		main		D										1		1	
	Snake Hole		sna		D								1	1	1	1	1	
Macumba	Carpamoongana	Mac	car	wh	D												1	
	Eringa		erin		D								1	1	1	1	1	1
	Ethawarra		eth		D								1		1			
	Andaranna		and		P								1		1			
	Winkies		win		P								1					
Neales	One Mile Bore	Nea	omil	bo						1	1							
	Old Peake Bore		opea											1				
	Michael Dam		mic	dm													1	
	Suspect Dam		sus														1	
	Three Sisters Dam		thr														1	
	Arckaringa Creek		ark	in							1							
	Hanns Creek		han								1							
	Lora Creek		lor								1							
	Ockenden		ocke								1							
	North Freeling Springs		fre	sp						1	1		1	1				
	Old Nilpinna		onil							1								
	Algebuckina		alg	wh	A	1	1	1	1	1	1		1	1	1	1	1	1
	Afghan		afg		D					1	1							
	The Cliff		clif		D					1								
	Cramps		cram		D	1	1	1			1						1	
	Hookeys		hook		D	1	1	1	1	1	1		1		1			
	South Cliff		sclif		D					1	1						1	
	South Stewarts		sste		D				1	1								
	Stewarts Waterhole		stew		D	1	1	1	1	1	1		1		1		1	
	Tardetakarinna Waterhole		tard		D					1								
	Baltacoodna		bal		P					1	1		1	1				
	Peake Crossing		pea		P	1	1	1	1	1	1		1		1			
	Warrarawoona Waterhole		war		P						1		1	1				
Kati Thanda–Lake Eyre South	Margarets Crossing	Eyr	marg	in											1			

Table 6-3: Floristic survey data in this study (BDBSA 2014)

Survey	Survey number	Date of visit	
		Start	*End*
Anangu Pitjantjatjara Yankunytjatjara Lands	23	14/10/94	30/10/98
Witjira National Park	25	06/07/91	27/08/97
Rare Rodents Project	48	30/05/92	28/05/03
Kowari Project	50	02/09/93	06/09/93
Stony Deserts	69	14/05/93	26/09/96
Arckaringa (ANZSES)	75	25/09/94	11/10/95
Kati Thanda–Lake Eyre South	77	03/07/95	24/07/96
Sandy Desert	94	17/06/96	19/08/06
Photopoints - NPWSA General	133	26/06/01	26/06/01
Dalhousie Survey by Scientific Expedition Group	161	21/07/03	23/07/03
Mt Willoughby Indigenous Protected Area	164	03/10/03	09/10/03
Wabma Kardarbu Mound Springs Inventory	175	10/05/04	15/05/04
Arid Rivers	207	30/04/05	30/09/05
Prominent Hill (Oxiana) Survey by Ecological Horizons	467	30/05/06	31/05/06
Woomera Prohibited Area	573	19/04/07	19/04/07
Finke IBRA Region Survey	605	20/05/08	26/05/08

Table 6-4: Hydrological data for the Neales–Peake catchment

No.	Name	Type	Longitude	Latitude	Bathymetry data	Water level data	Seepage loss
1	Peake Bridge/Crossing	WH	135.806	-28.0386	Yes	Yes	groundwater-fed no loss
2	Algebuckina Waterhole	WH	135.829	-27.892	Yes	Yes	not leaky
3	South Stewart Waterhole	WH	135.380	-27.699	Yes	Yes	leaky (3.85 mm/day)
4	Afghan Waterhole	WH	135.346	-27.447	Yes	Yes	not leaky
5	Angle Pole	WH	135.398	-27.500	Yes	-	not leaky
6	Shepherds	WH	135.424	-27.559	Yes	-	not leaky
7	Hookey	WH	135.433	-27.596	Yes	-	not leaky
8	Mathieson	WH	135.404	-27.660	Yes	-	not leaky
9	Stewart	WH	135.381	-27.688	Yes	-	leaky (3.85 mm/day)
10	Cramps Camp	WH	135.382	-27.700	Yes	-	leaky (3.85 mm/day)
11	Fish Hole	WH	135.261	-27.745	Yes	-	not leaky
12	Hagans Hole	WH	135.279	-27.867	Yes	-	not leaky
13	South Cliff	WH	135.995	-27.910	Yes	-	not leaky
14	Cliff	WH	135.986	-27.891	Yes	-	not leaky
15	Tardetakarinna Waterhole	WH	136.136	-28.016	Yes	-	groundwater-fed no loss
16	Warrawaroona	WH	135.912	-28.047	Yes	-	not leaky
17	Baltacoodna	WH	135.908	-28.037	Yes	-	groundwater-fed no loss
18	Cootanoorina	WH	135.306	-28.175	Yes	-	not leaky
19	Birribiana	WH	135.254	-28.2190	Yes	-	not leaky
20	North Freeling	WH	135.892	-28.053	Yes	-	not leaky
21	mid Peake	Logger	135.558	-28.107	-	Yes	-
22	Nilpinna Creek	Logger	135.698	-28.050	-	Yes	-
23	Lora Creek	Logger	134.944	-28.318	-	Yes	-
24	Arckaringa Creek	Logger	135.082	-28.122	-	Yes	-
25	Ockenden Creek	Logger	135.741	-27.840	-	Yes	-
26	Tardetakarinna channel	Logger	136.152	-28.026	-	Yes	-
27	Lambing Creek	Logger	136.306	-28.227	-	Yes	-
28	South Branch of Neales	Logger	135.123	-27.506	-	Yes	-
29	Mid Neales	Logger	135.588	-27.853	-	Yes	-
30	Retard	Logger	136.151	-28.022	-	Yes	-

Table 6-5: Hydrological data for the Macumba catchment

No.	Name	Type	Longitude	Latitude	Bathymetry data	Water level data	Groundwater connectivity
1	Eringa Waterhole	WH	134.723	-26.286	Yes	Yes	not leaky
2	Ethawarra Waterhole	WH	134.640	-26.774	Yes	-	-
3	Carpamoongana WH	WH	134.763	-26.755	Yes	-	-
4	Ekeetatrinna Waterhole	WH	134.802	-26.321	-	-	groundwater-fed?
5	Hamilton Creek	Cross	135.291	-26.540	-	-	leaky?
6	Algebuckina Waterhole	WH	135.819	-27.200	-	Yes	-

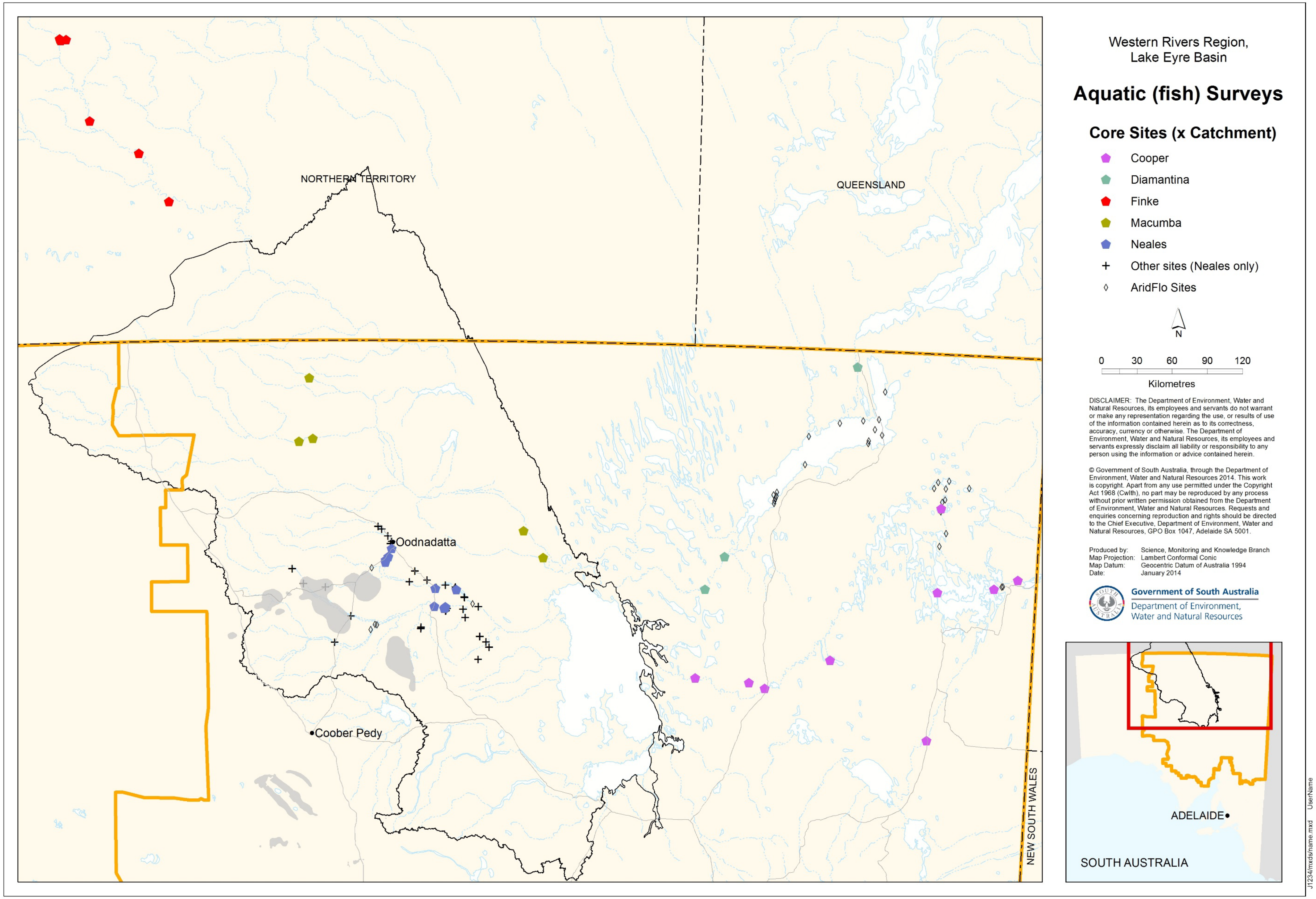


Figure 6-1: Aquatic survey and monitoring sites

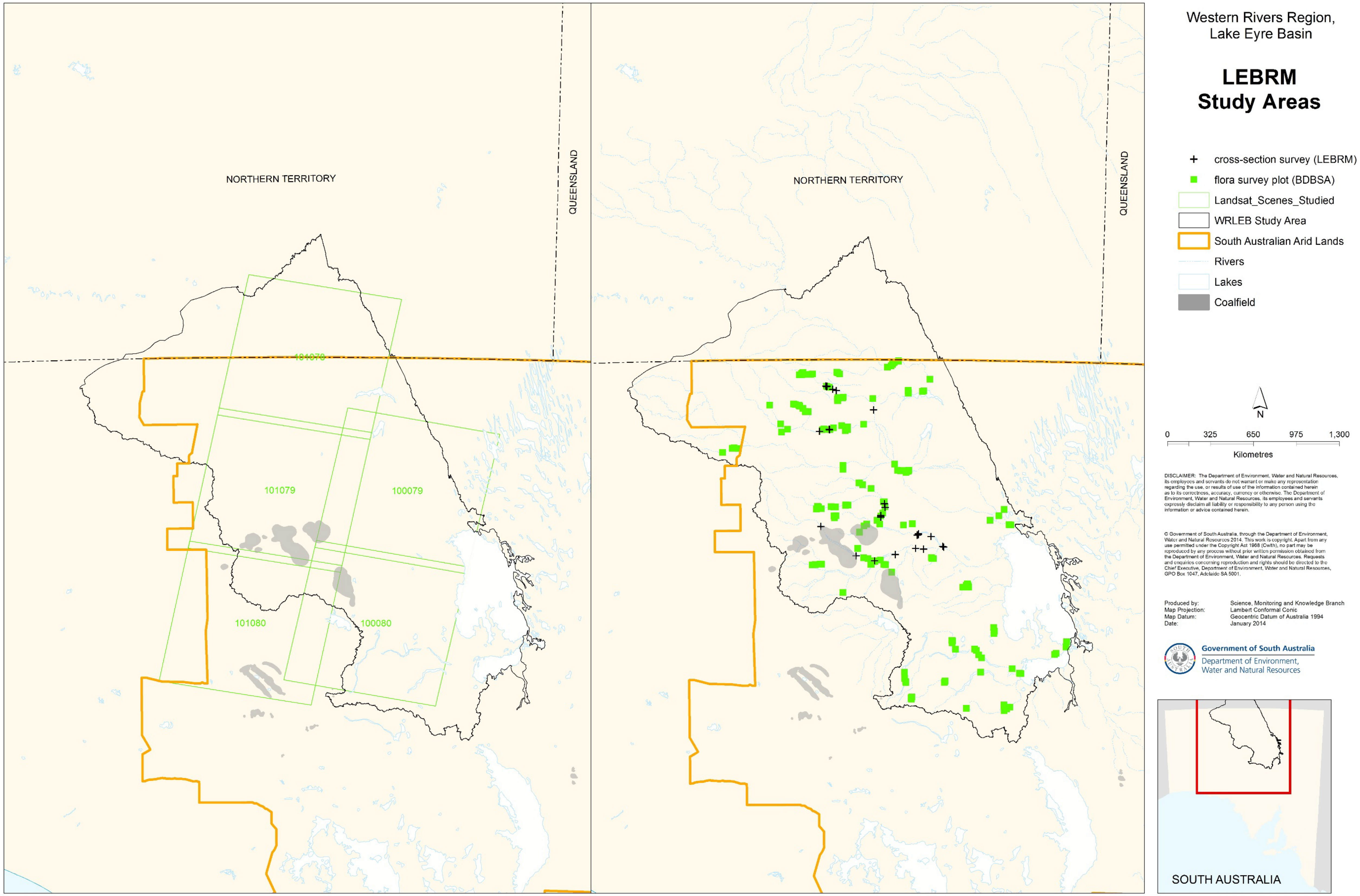


Figure 6-2: Extents of satellite imagery scenes and additional survey sites

B. Field survey methodology

B.1 Hydrological monitoring and surveys

Methods for hydrological monitoring and surveys are outlined in Costelloe et al. (2011). For details on the Neales–Peake rainfall-runoff model, see Montazeri and Osti (2014). Sap-flow monitoring methods are in Ryu et al. (2014).

B.2 Fish and water quality monitoring

Methods for fish and water-quality sampling (Schmarr et al. 2014) follow those used for LEBRA (McNeil and Schmarr 2008; McNeil et al. 2009; Cockayne et al. 2012) and Critical Refugia sampling (McNeil et al. 2011).

At each site, substrate type, instream structure, rate of flow and connectivity to the main channel were assessed. Percent cover of aquatic, emergent and riparian macrophytes was estimated and dominant species identified (Sainty and Jacobs 2003). A point of maximum depth was identified at each site where water quality was recorded.

Water quality parameters – dissolved oxygen, water temperature, pH and salinity – were analysed on-site using a Horiba multi-parameter meter. Measurements were taken at the water surface and at 50 cm depth intervals and used to create a vertical profile, revealing mixing (e.g. stratification) in the water column. Water at highly saline sites was diluted to measure within the limits of the water-quality meter, then back-calculated to full strength.

Each site was sampled using a standard set of fyke nets with six small fyke nets and two double-wing large fyke nets, consistent with surveys for the Lake Eyre Basin Rivers Assessment (McNeil and Cockayne 2010). The characteristics of the nets were:

- Small fyke: meshed single-winged design (3-m wing, 4-mm mesh, 3-m funnel, 0.6 m high), effective in sampling small bodied fish in shallow water, and
- Double-wing large fyke: meshed double-wing design (2 x 10 m wing, 12-mm mesh, 5-m funnels, 1.2 m high), used for large-bodied fish and set in deeper water.

All nets were set within 250 m of an access point (usually the GPS waypoint). Small fyke nets were set in shallow locations targeting microhabitats within the waterhole (e.g. complex snags or dense stands of submerged vegetation). Double-wing large fykes were deployed paired and in opposition in deeper water, targeting larger fish. Additional *ad hoc* sampling used visual surveys and nocturnal spotlighting to supplement presence/absence data.

Fyke nets were anchored using heavy gauge chain clipped to the cod and wing ends. Wing tips were tied off on natural structures or onto stakes. Two polystyrene buoys were placed in the cod end to force a pocket of net above the surface. This created a space where bycatch (birds, water rats, turtles) could breathe until the net was processed. The nets were set before dusk, left overnight and collected after dawn, for a minimum of 14 hours. This allowed capture during crepuscular activity.

The following outlines processing procedures, adapted from McNeil and Cockayne (2010):

- All fish were identified using well-known keys (Wager and Unmack 2000; Allen et al. 2002; J Pritchard unpublished).
- The lengths of the first 100 individuals of each species were measured (total length, mm).
- Measured fish were visually inspected for disease (e.g. lesions, scoliosis) and spawning condition, and returned to the water at the point of capture. Voucher specimens were kept where identification was uncertain.
- Fish not measured were counted.

Records were also kept of bycatch, including Yabbies (*Cherax destructor*), Shrimp (*Macrobrachium, Paratya*, *Caridina* spp.) and Freshwater Crabs (*Austrotelphusa transversa*).

B.3 Vegetation transects and surveys

Floristic data were queried from BDBSA (Biological Databases of South Australia) for the search area in Figure 6-3 (details are in Table 6-3). Methods followed the DEH (1997) manual for vegetation surveys. Summary metadata are at: http://www.envapps.sa.gov.au/emap/envmaps-query.do?cmd=su.SurveySummaryMain

Figure 6-3: Floristic surveys and plot locations

B.4 Additional surveys

Three survey techniques were employed in May 2013 and February 2014 transect surveys of Neales–Peake and Macumba waterholes. Results are in Section 3.4.5 and Appendix E.

1. Traverses (transects)

Floodplain and channel traverses were surveyed using a Differential Geographic Positioning System (Trimble® Real-Time Kinematic satellite navigation). Traverses were surveyed perpendicular to the slope across floodplain and channel, recording the coordinate and elevation of points to demarcate change in slope, features of interest and major vegetation and landform transitions, with density of points varied according to the nature and complexity of transitions. Dominant species and or landform descriptors were recorded for the zone of interest.

Additional detailed topographic surveys were undertaken in February 2014 using a Topcon® Total Station. The surveys undertaken were used to supplement data collected in May 2013 and for additional sites of note.

Table 6-6: Sample results from the lower elevation (metres ±SE) of vegetation-zone boundaries relative to top of bank (Macumba sites)

Site	RRG*	Gidgee	f/p-terr	Replication
Eringa	-1.2 ± 0.11	0.2 ± 0.10	1.1 ± 0.11	*n* = 15, 18 and 9 respectively
Carpamoongana	-1.4 ± 0.19	0.3 ± 0.40	0.9 ± 0.09	*n* = 18, 6 and 7 respectively
Ethawarra	-0.9 ± 0.27	0.5 ± 0.10	0.7 ± 0.11	*n* = 4, all
All data	**-1.26 ± 0.017**	**0.27 ± 0.016**	**0.92 ± 0.015**	***n* = 38, 33 and 21 respectively**

*RRG = River Red Gum; **f/p/terr = floodplain terrace transition (i.e. outer floodplain boundary)

2. Random sub-transects

Sub-transects were surveyed to record variation in bank heights, channel slope and overlapping vegetation distributions (commonly observed at Neales sites). Ten randomly located sub-transects (minimum 5-m spacing) were run perpendicular to 100-m longitudinal transects along the top of levee. The 100-m transects were located either centrally for small waterholes or divided the waterhole into thirds at larger sites. The aim was to provide adequate elevation data replication to estimate variation in transitions of vegetation and stream morphology.

3. Tree size distribution (preliminary)

Transects at two sites (Hookeys, Shepherds) were surveyed to quantify the size distribution of common canopy species in the floodplain to investigate whether discrete age cohorts could be identified. Tree diameter at breast height (1.3 m) was recorded in longitudinal transects covering the bank to 10 m from the natural channel levee.

Waterholes had very few recruits or juveniles in the riparian zone (defined as the top of levee slope to 1 m below cease-to-flow level), prompting questions as to whether populations were self-replacing. Data suitable to determine tree size distributions were sought. All trees within a 100-m linear transect along the watercourse were measured for circumference at breast height (1.3 m). All trunks or branches at this linear distance from the base of the tree were measured (whether vertical or horizontal or in between).

For each tree, the chainage along the linear transect, lateral location (three classes: bank, top of bank or levee) and the number of trunks/branches at 1.3 m were recorded. Stems with a diameter < 50 mm were measured as a diameter using a steel builder's tape directly; larger branches and trunks were measured by taking the circumference using a fibreglass or metal builder's tape. Any evidence of wood cutting was noted, with the number and dimensions of the removed branch measured at the cut. All data were converted to diameter and binned for presentation.

Emerging seedlings were observed at most sites. These were a few days to weeks old and typically <100 mm tall. Taller seedlings had been grazed to near-ground level and were beginning to recover leaves and stems.

Figure 6-4 shows distribution data from two sites, binned into 100 mm-size classes. Distinct distributions can be discerned, with trees at Shepherds waterhole predominantly in the 10–200 mm diameter class, while trees from the Hookeys sites were predominantly between 500–600 mm. Only one tree smaller than 300 mm was identified at Hookeys.

Tree density varied between sites: at Hookeys 33 trees were observed in 200 m total sample, while 89 trees were observed over 150 m at Shepherds. Larger size classes were similarly distributed, while Hookeys had more trees in mid-size classes and Shepherd in smaller sizes (Figure 6-5).

The sites are around 5 km distant on the same reach of the Neales River near Oodnadatta. Both sites are open to the public, but Hookeys is located on the main road between Oodnadatta and Coober Pedy (the Kempe Road) and is heavily visited, while Shepherds is located several kilometres behind the Oodnadatta township on the Town Common and is less accessible. Grazing pressure appears the most likely factor affecting size class distribution – Shepherds is not grazed but Hookeys is. Grazing also presents one possible explanation for the dominance of middle-sized trees in the Hookeys data – with lower competition from juvenile cohorts, trees established at the onset of grazing were able to grow more rapidly. As growth rates slow with age, larger trees may have been fully grown at the time of grazing commencing and have maintained similar size distributions despite changes in grazing pressure.

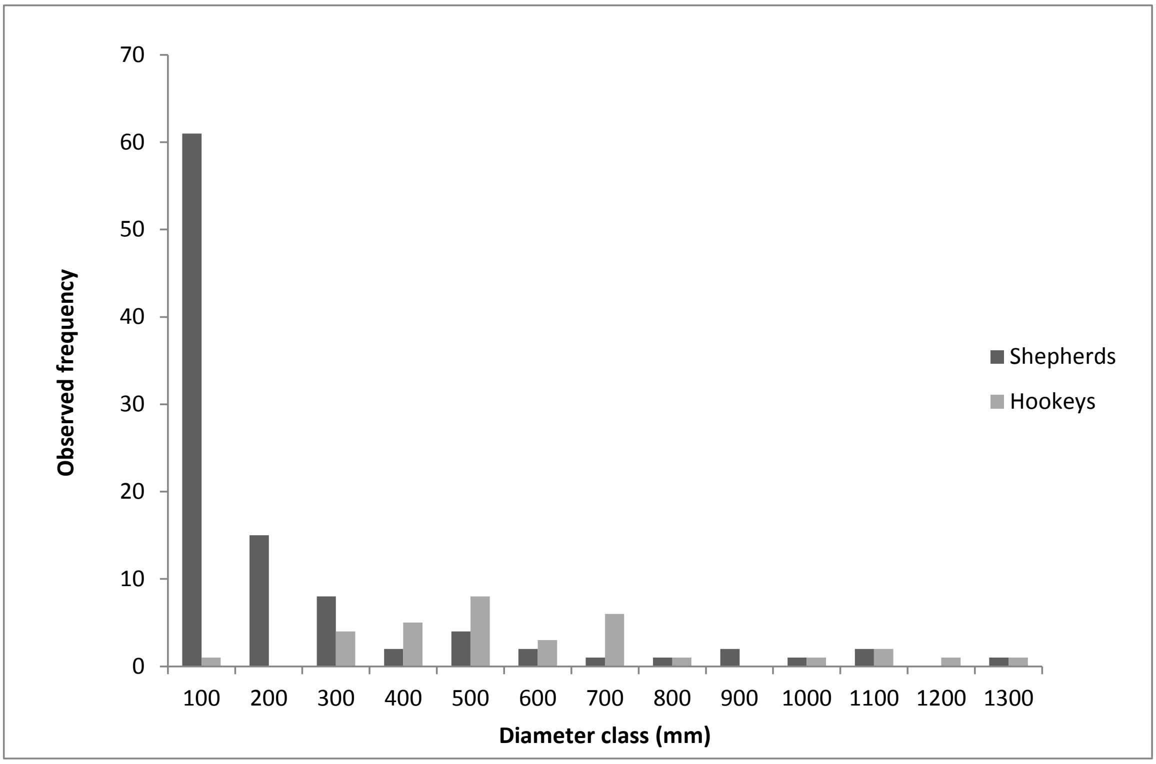


Figure 6-4: Size-class distribution of Coolabahs at the waterhole and floodplain levee (within 10 m from top of bank) at sites in the Neales River

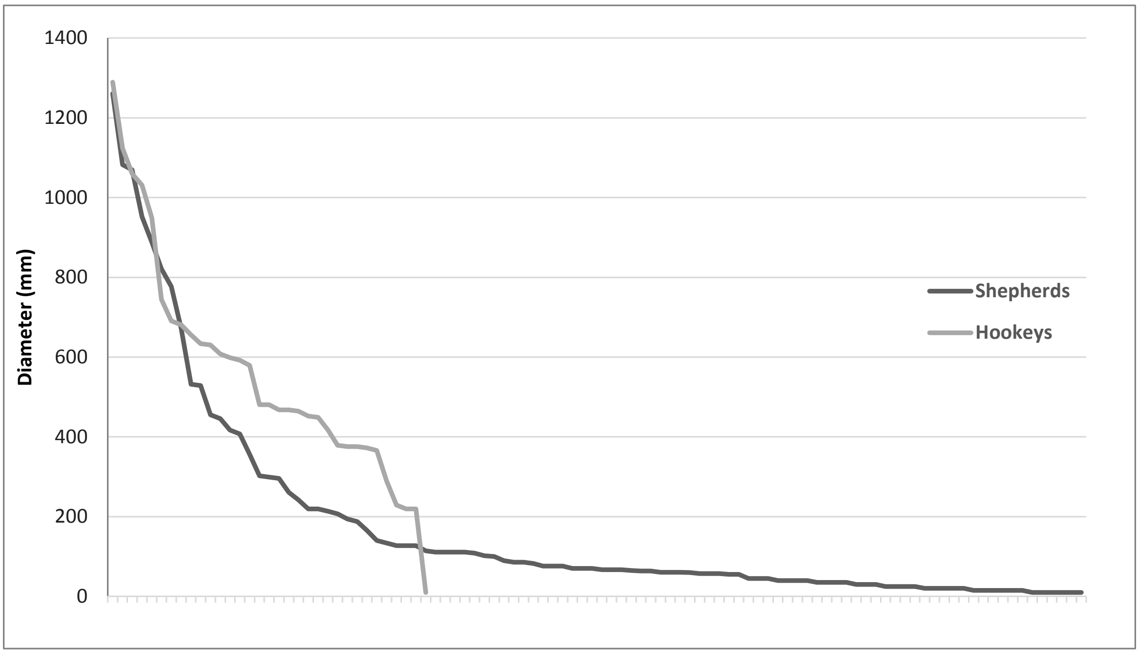
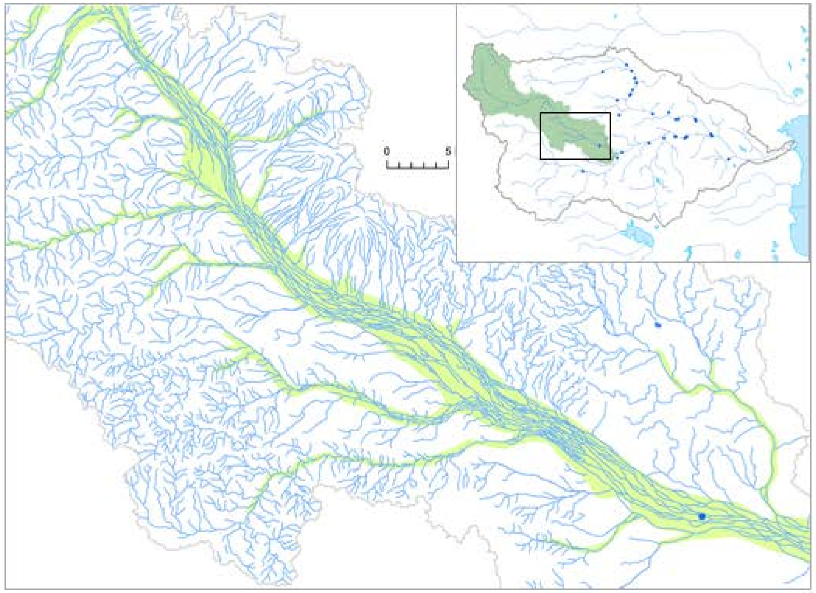


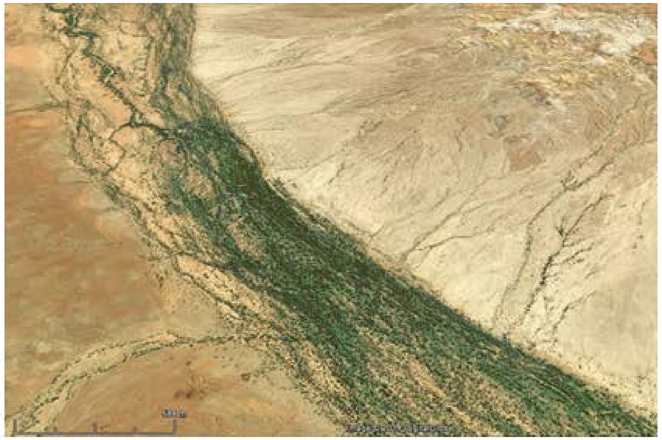
Figure 6-5: Ranked tree size distributions (all data)

Larger trees are distributed almost identically; intermediate trees are larger at Hookeys and smaller trees are absent. Mid-aged trees may perform better as a result of less competition for resources, particularly soil moisture.

C. Representative reaches of the Neales–Peake catchment

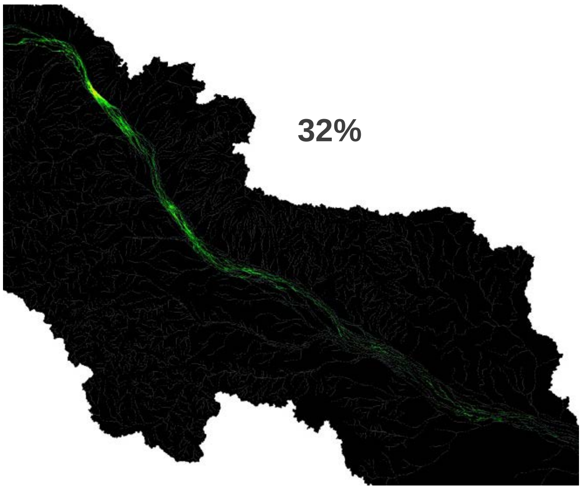
C.1 Arckaringa Creek



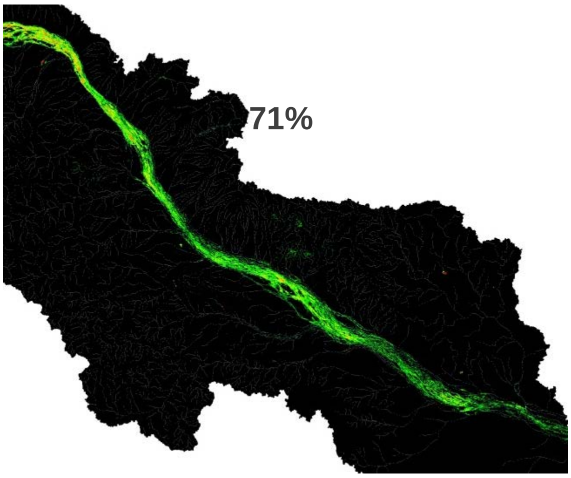


C.2 Arckaringa Creek - Flood inundation model inputs

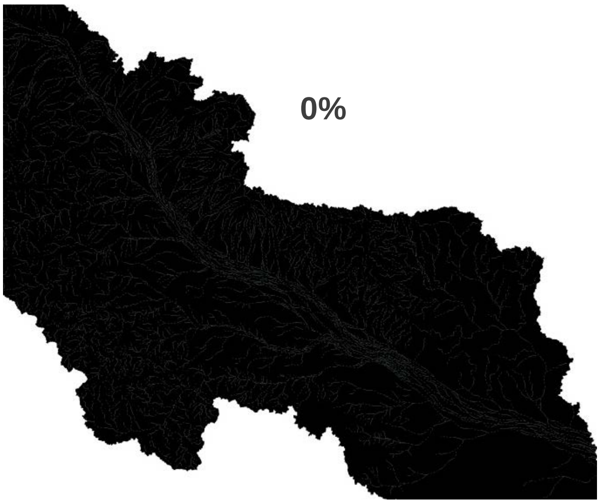
March, 1999



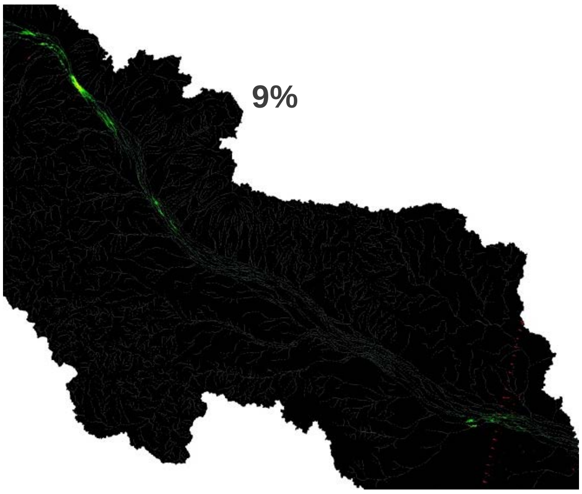
May, 2000



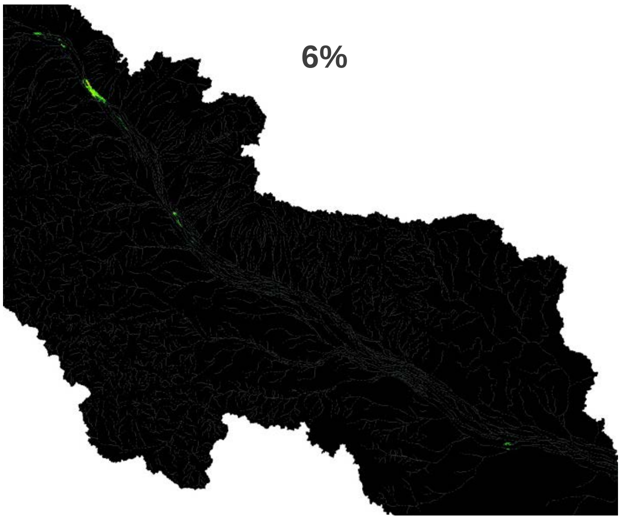
January, 2001



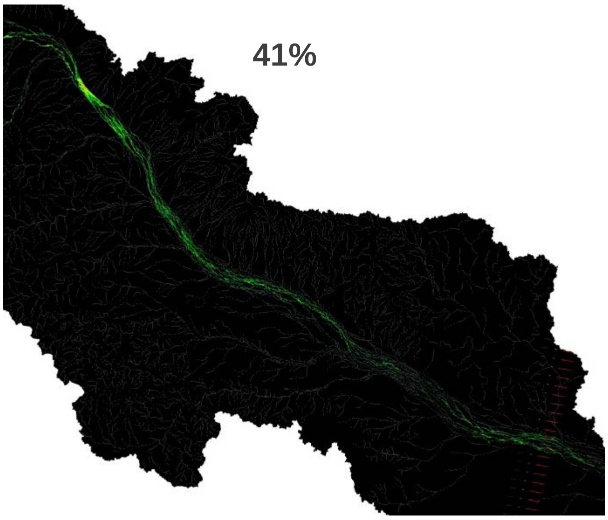
September, 2001



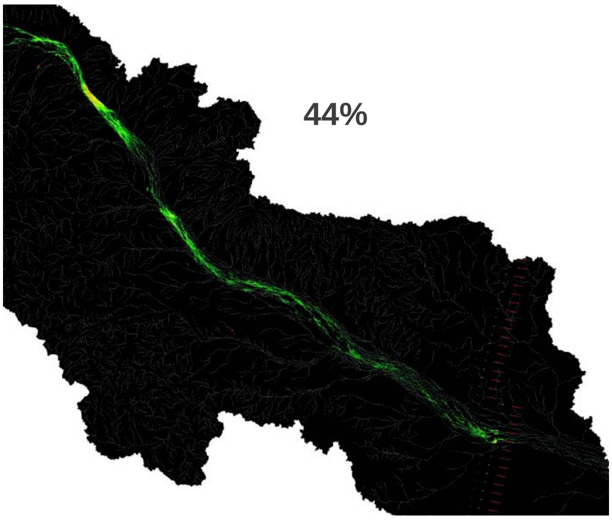
March, 2003



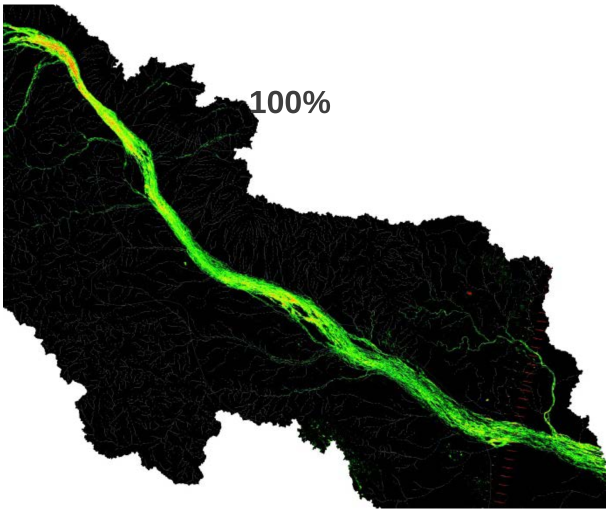
March, 2008



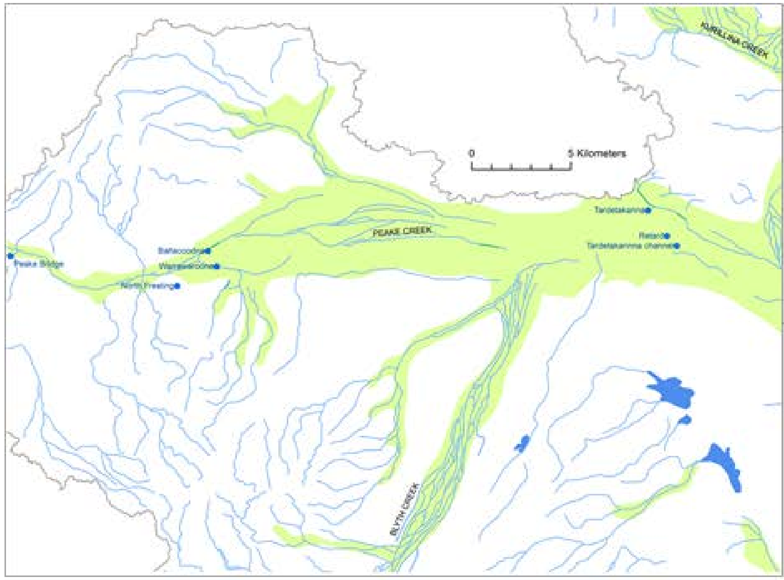
May, 2010

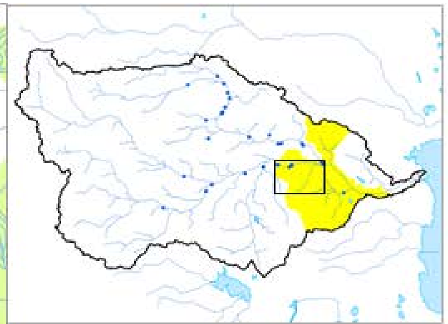


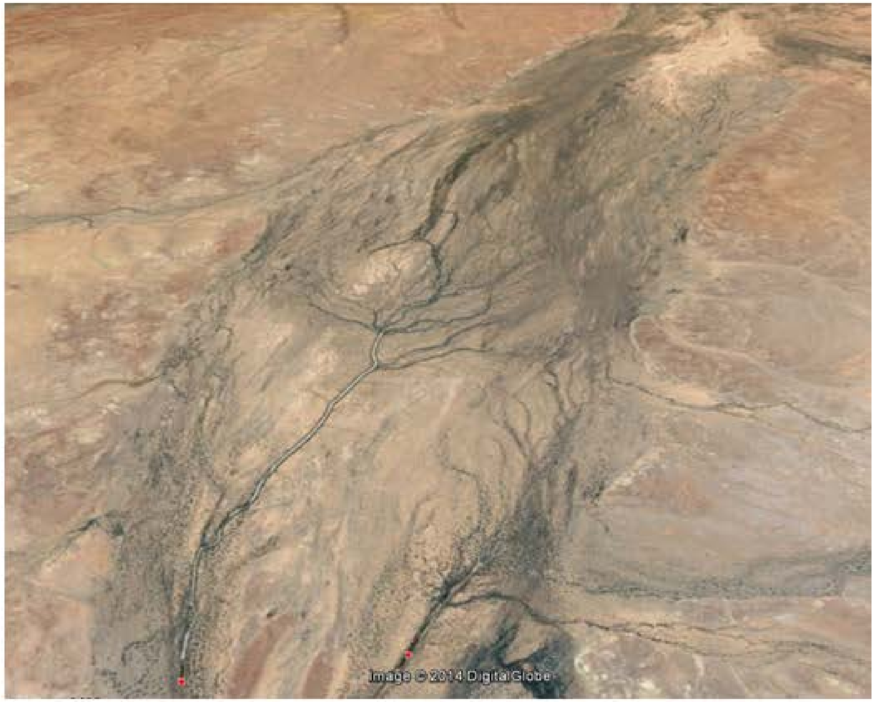
April, 2011



C.3 Peake-confluence reach

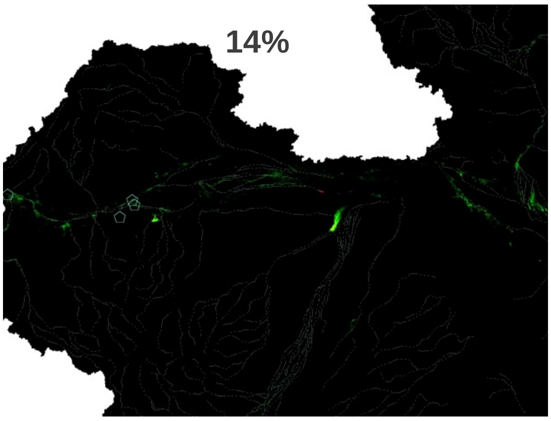






C.4 Peake–confluence – Flood inundation model input

March, 1999



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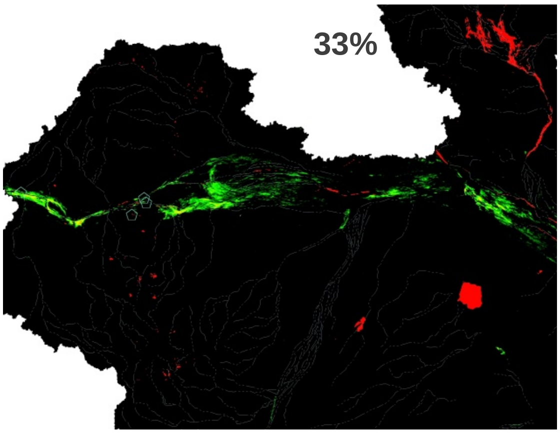
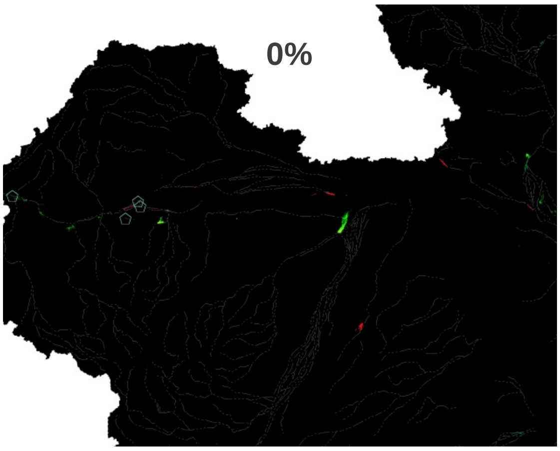


January, 2001



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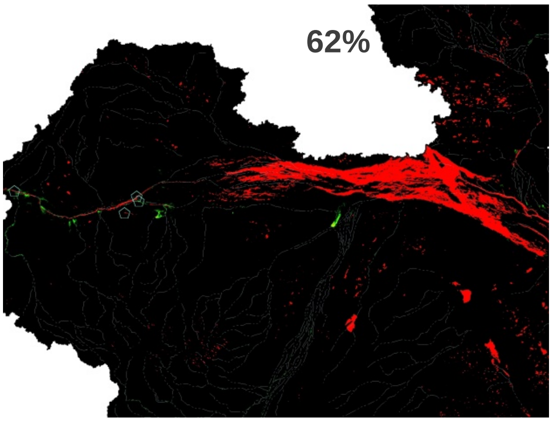
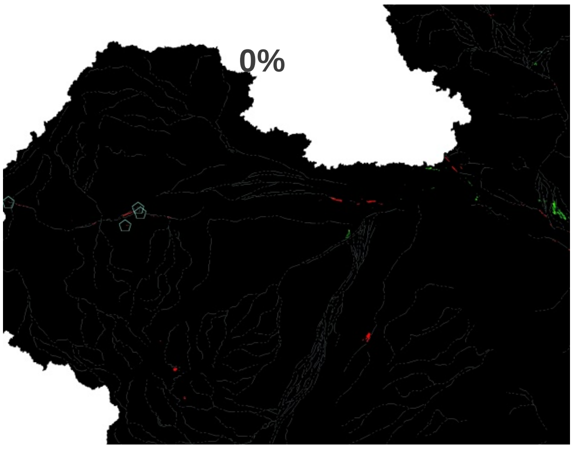
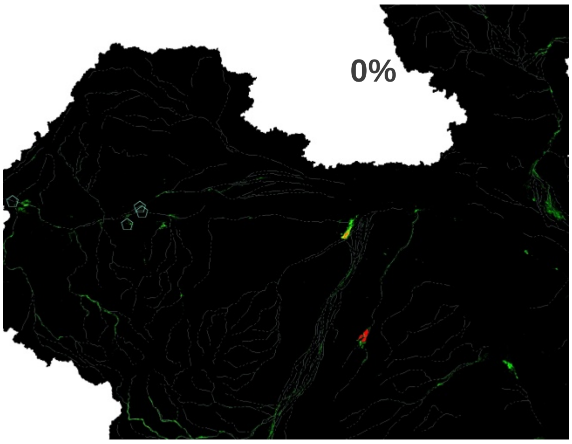


March, 2003



March, 2008



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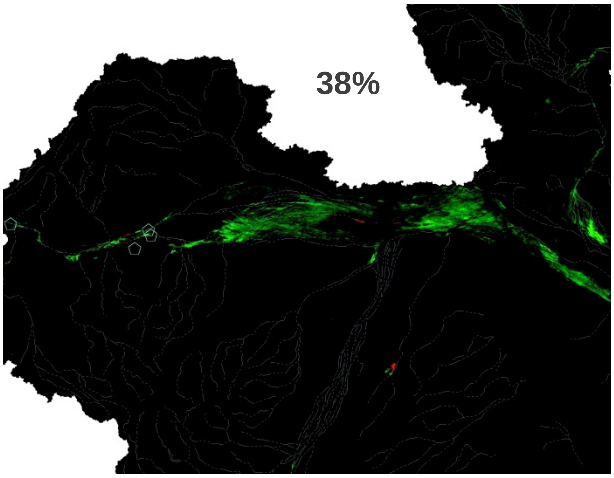
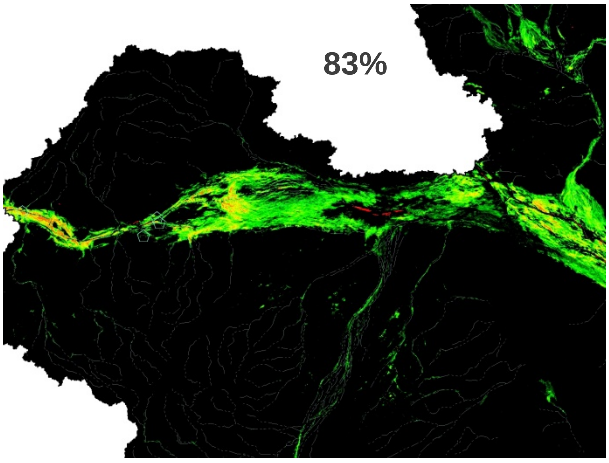
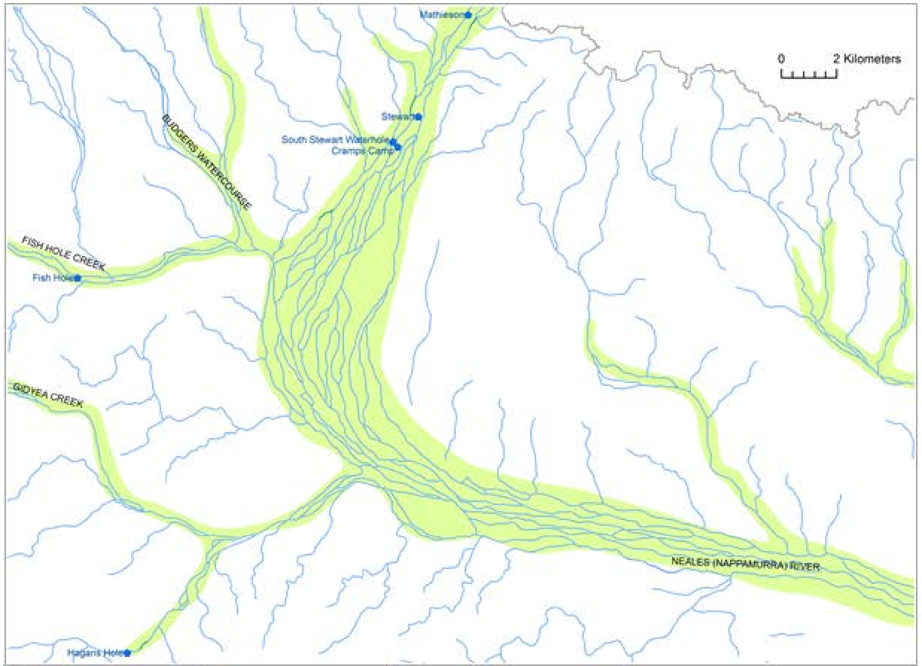
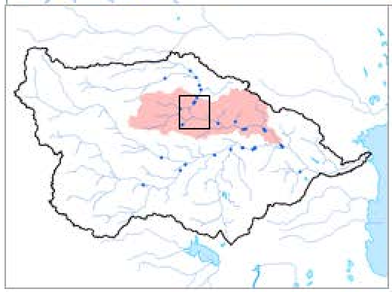


April, 2011



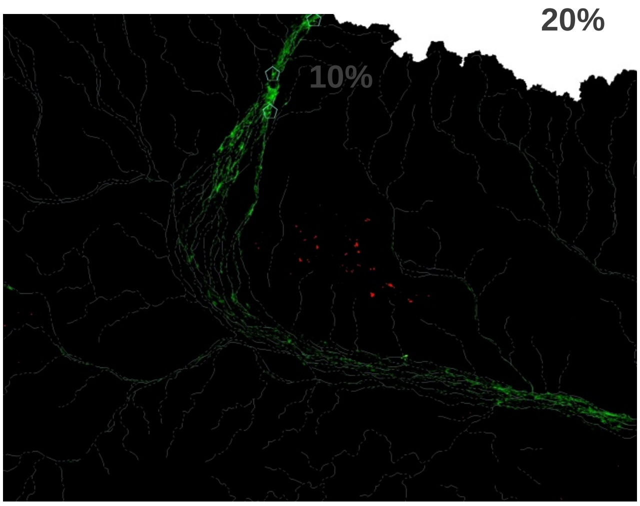
C.5 Stewarts–Neales Corner reach



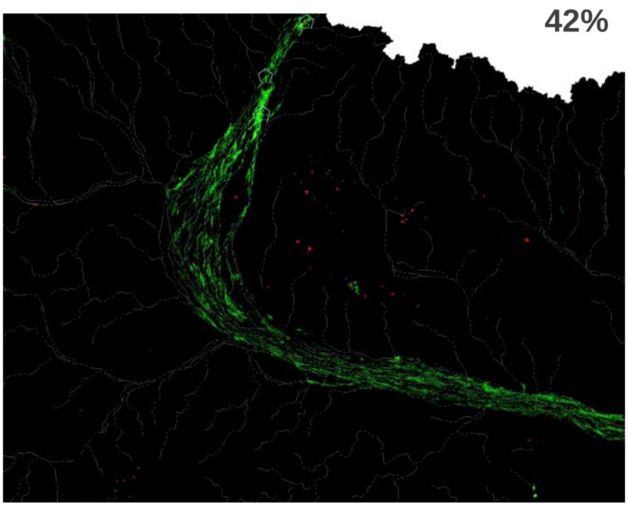


C.6 Stewarts–Neales Corner – flood inundation model inputs

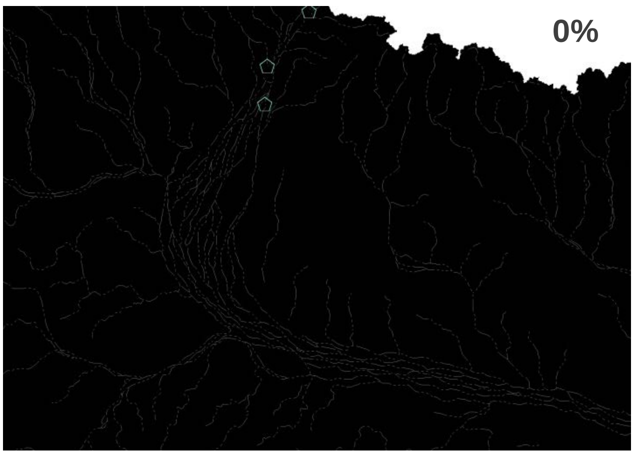
March, 1999



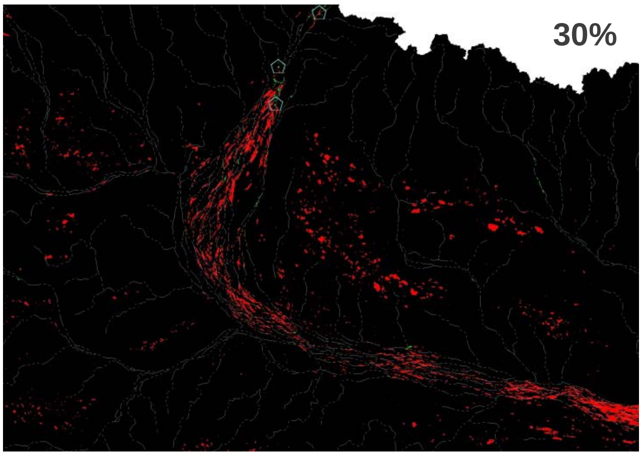
May, 2000



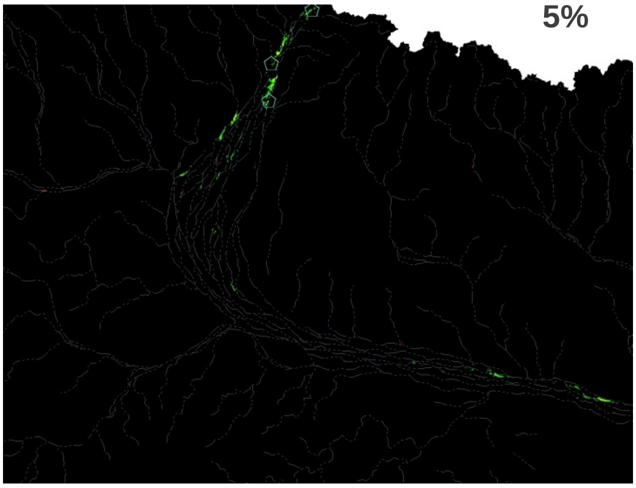
January, 2001



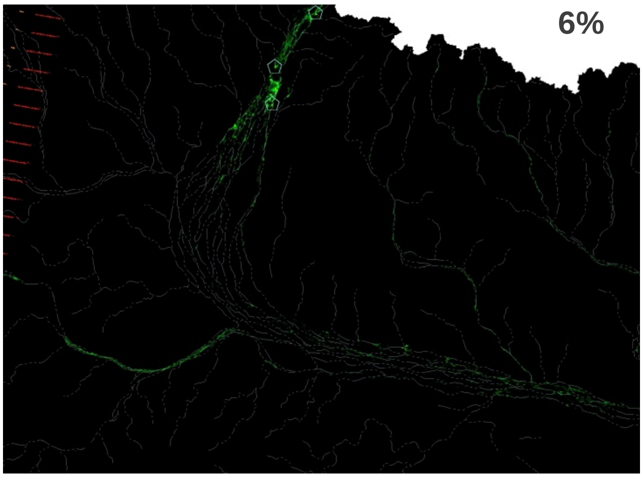
September, 2001



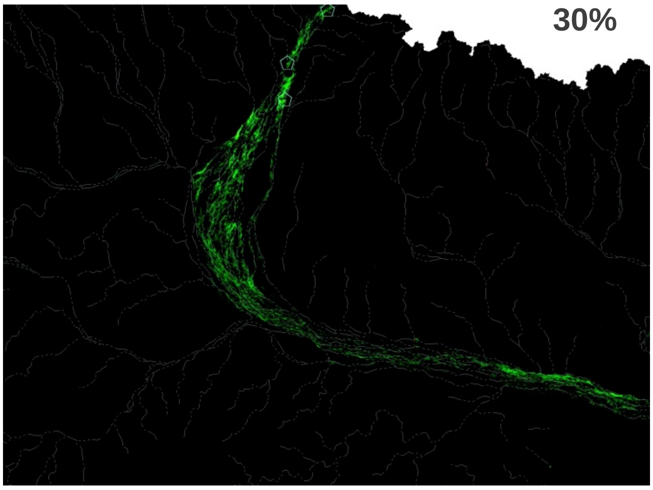
March, 2003



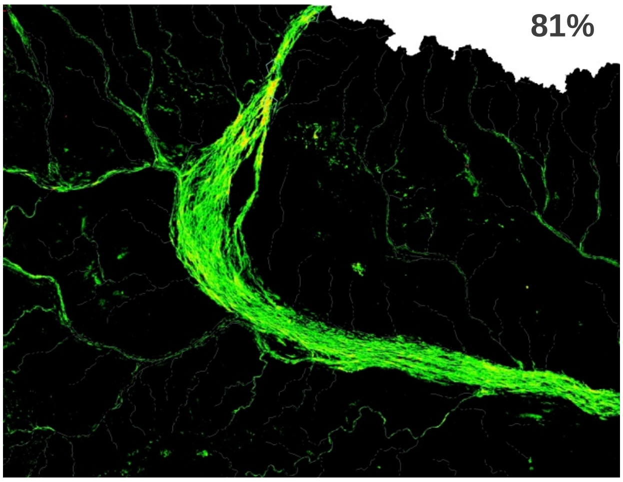
March, 2008



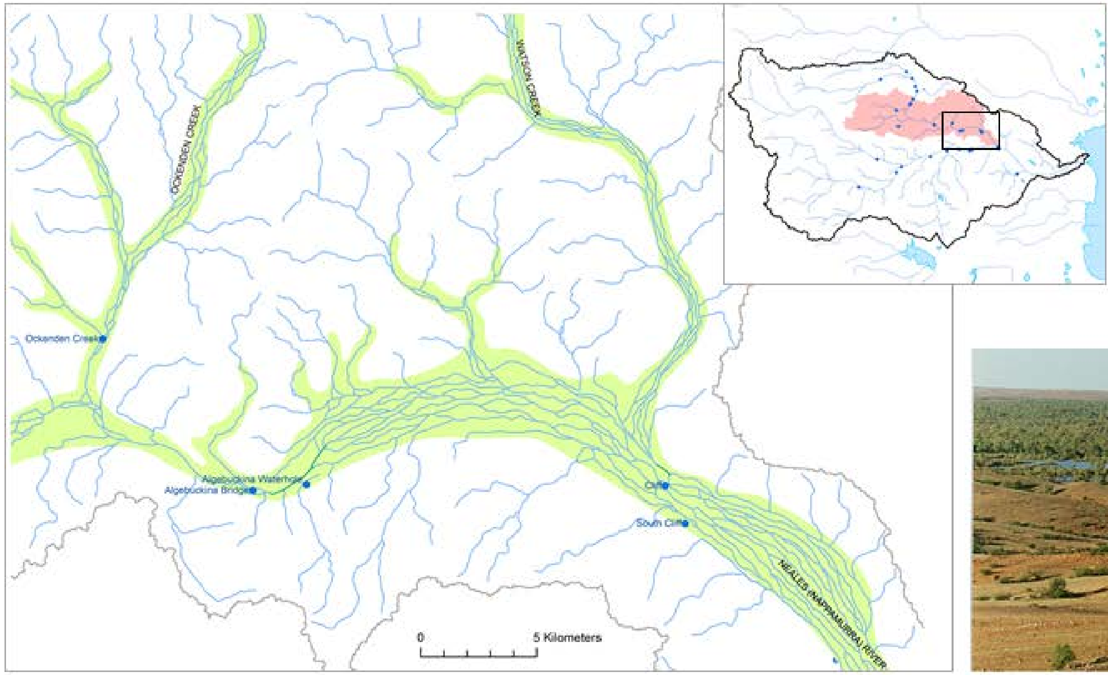
May, 2010

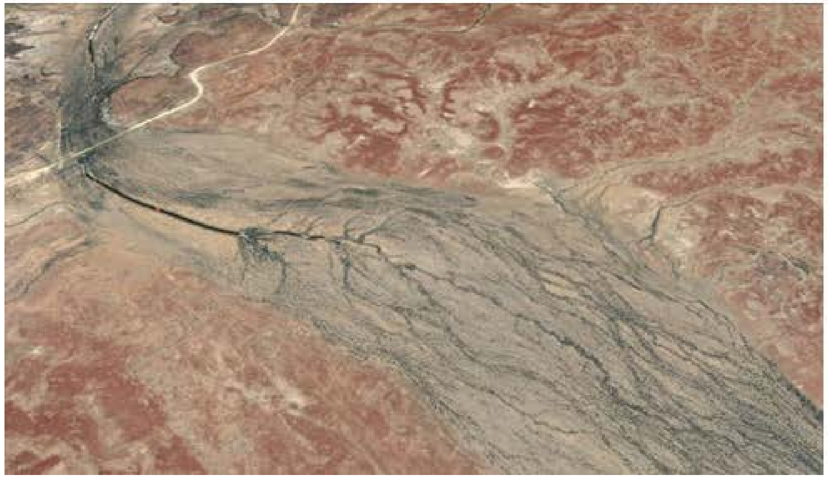


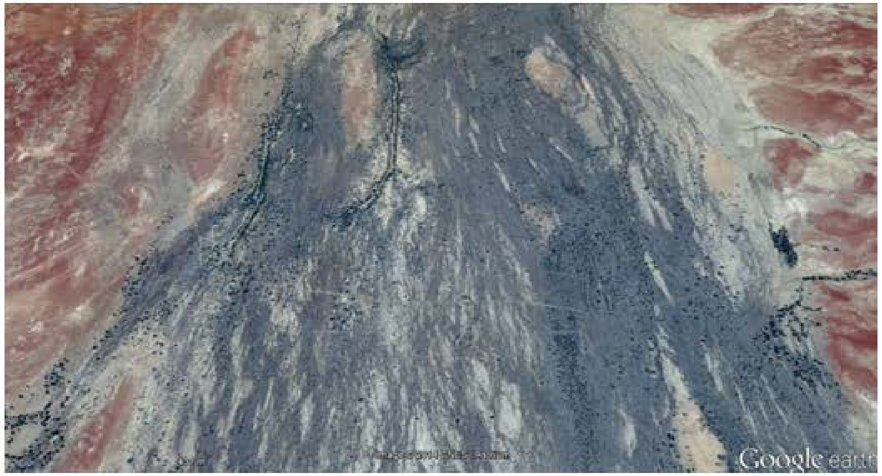
April, 2011



C.7 Algebuckina–Tardekarinna reach







C.8 Algebuckina–Tardekarinna – Flood inundation model inputs

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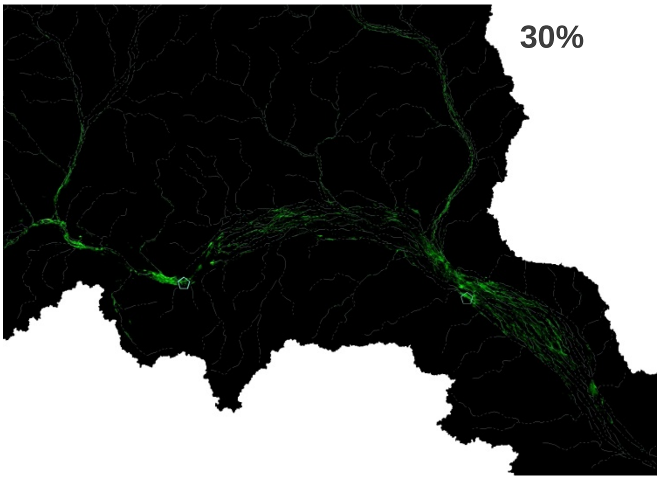
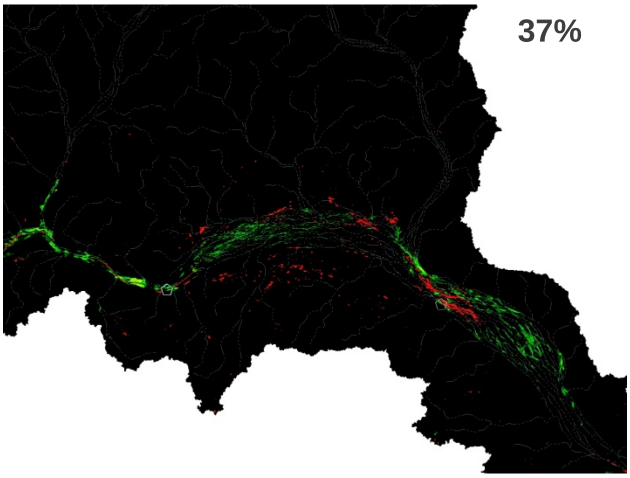
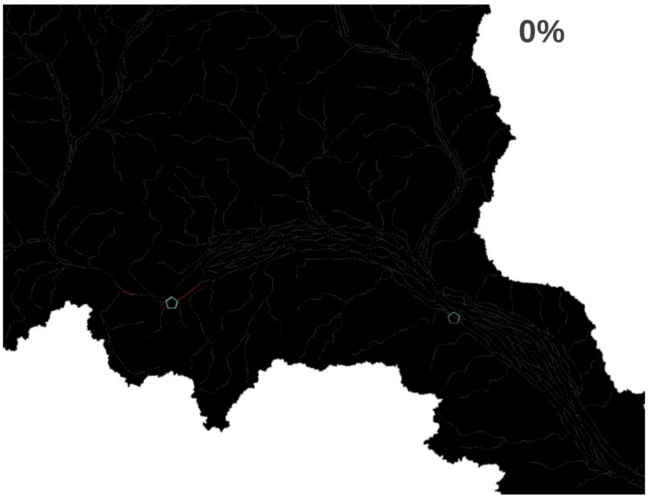


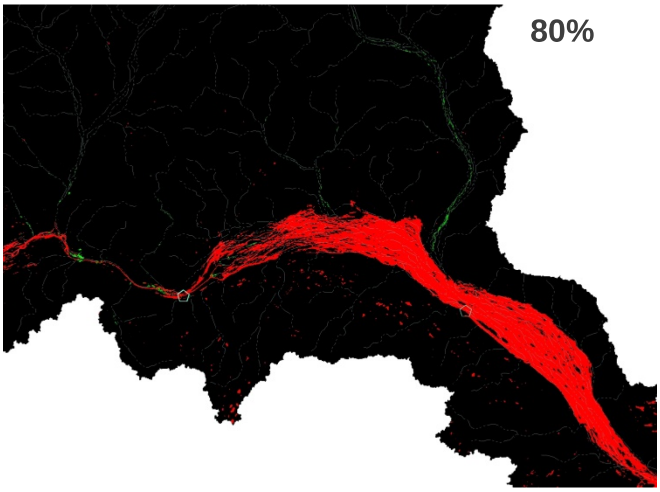
May, 2000



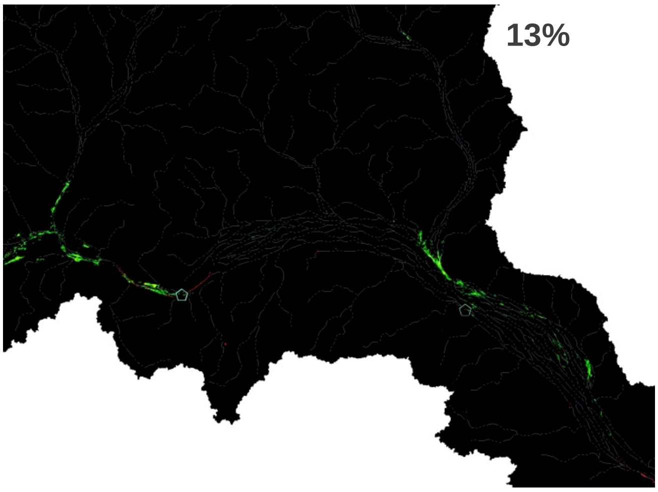
January, 2001



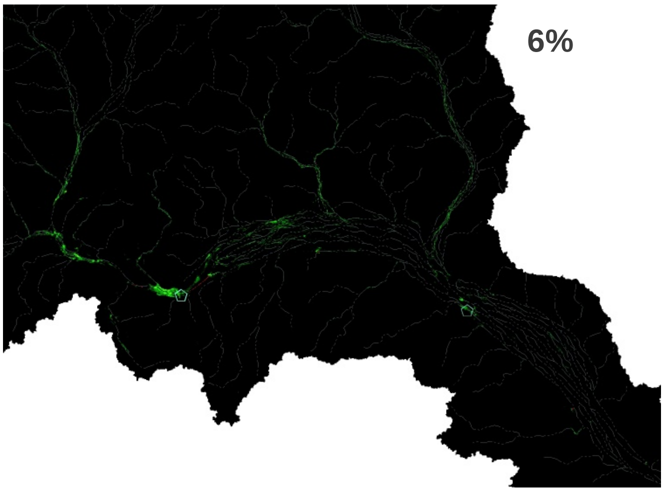
September, 2001



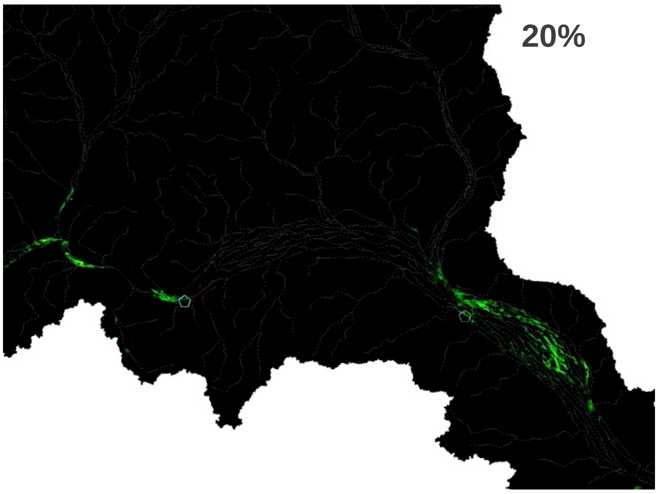
March, 2003



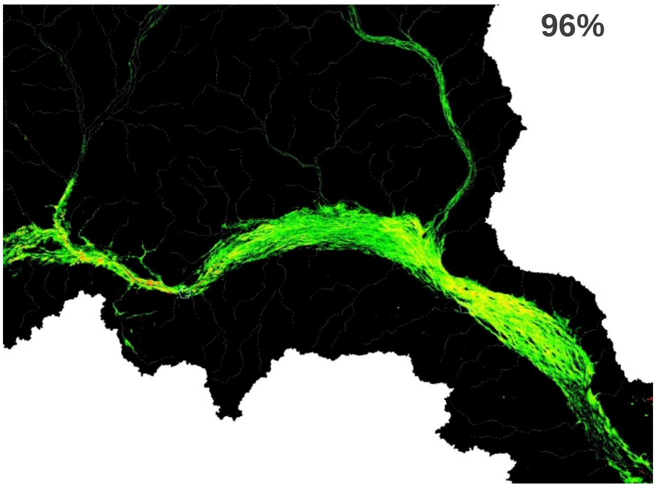
March, 2008



May, 2010



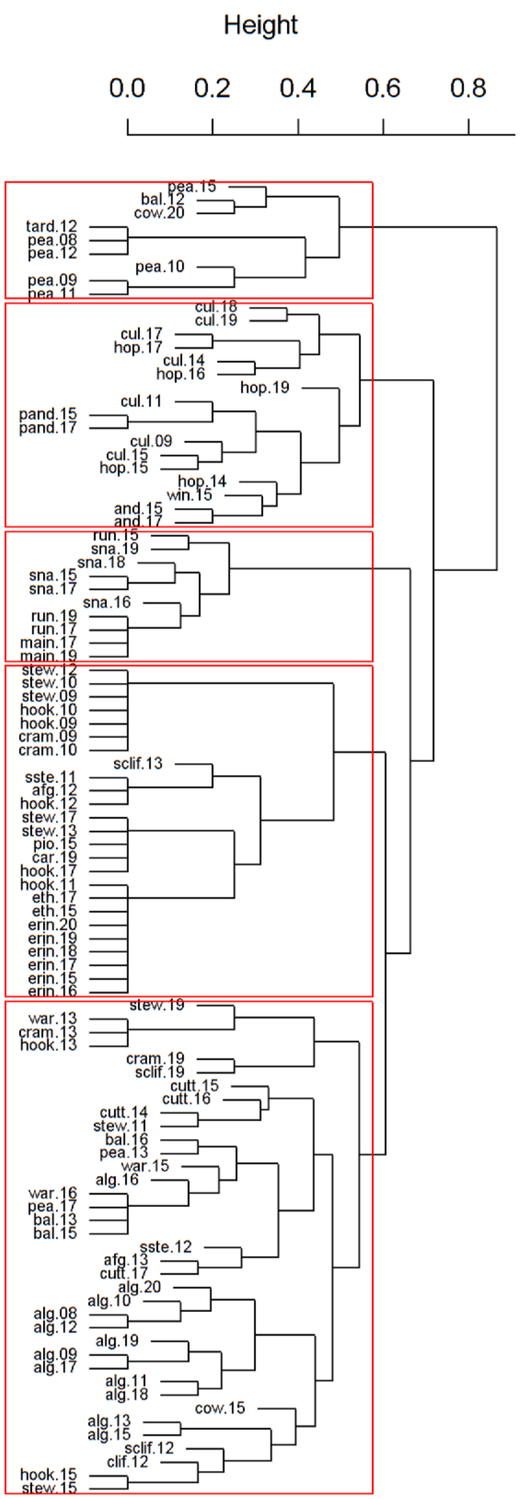
April, 2011



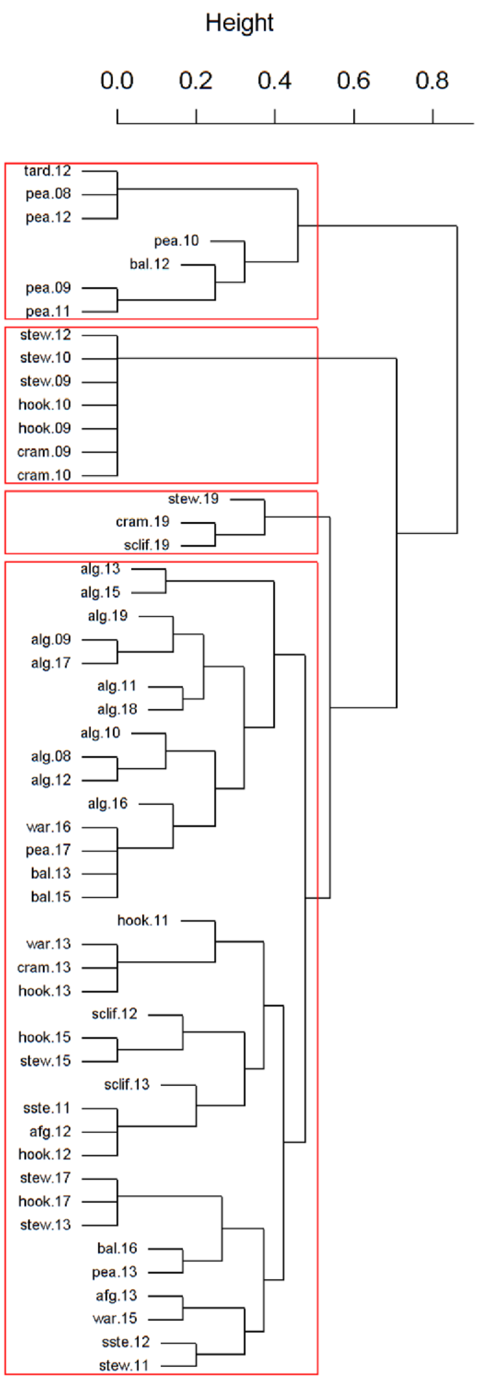
D. Statistical analyses

D.1 Fish groups (labels are site.visit)

i. Catchment-scale



ii. Patch-scale



D.2 Flow variability axes

Defining flow regimes is challenging, particularly in the arid zone, because of poor spatial resolution of gauge data and event-event variability. Analyses using modelled streamflow data were used as a guide (Olden et al. 2012; Montazeri and Osti 2014). The analysis undertaken reveals flow metrics are generally clustered along six main axes of variation[17] describing flow, no flow and permanency facets of the hydroecology (Table 6-7). Axes are defined by a number of flow metrics, together describing frequency (Ff), duration (Fd) and magnitude (Fm) of flows, drying (Dr) and drought (Dd) conditions, and water levels (Wl).

Table 6-7: Flow axes and indicator metrics for WRLEB natural flow regimes

Flow axes			**Principal component axes***						**Indicator metrics**
Tier 1	*Tier 2*		PC1	PC2	PC3	PC4	PC5	PC6	
Flow	Frequency	Ff	0.43	**-0.72**	-0.11	**-0.38**	0.28	-0.09	qve.90, qve.730
	Duration	Fd	**1.00**	-0.15	**-0.29**	-0.11	-0.15	0.08	qd.90, qs.rlm.365
	Magnitude	Fm	**0.81**	0.32	-0.02	0.11	0.04	0.10	q.spell, qv.90, qve, qve.spell, qd.max.yr, qs.rlm.max.yr, qs.fal.max.yr
No flow	Dry run	Dr	-0.70	0.13	0.00	-0.24	-0.33	0.38	zqd.spell, zwld.low.spell
	Drought	Dd	-0.69	0.00	**-0.16**	-0.27	-0.35	0.38	zwld.low.365, zqd.max.yr
Permanency	Water level	Wl	0.46	**-0.69**	0.15	**0.40**	-0.19	0.11	wl, wl.st, wh.v

*Bolded numbers refer to significant relationships between PC scores and flow axes (Figure 6-6)

Figure 6-6 shows plots of the first five principal components (PCs) that together represent 80% of the hydrological variation in Neales catchment refuge waterholes. The relationships between PC axes, flow axes and site visits were used to decompose the variation into four facets : flow stage (PC1), flow intermittency (PC2), water level stability (PC3) and time since inundation (PC4). Flow stage (PC1) separated sites under flow and no-flow conditions, with many sites deviating from their no-flow baseline during the 2010–11 floods, and a marked increase in flow duration and magnitude associated with catchment-wide connecting flows (Figure 3-11).

The second to fourth PCs describe facets of the hydrological regime independently of hydroclimatic conditions, as seen in the correlation between flow and no-flow Tier 2 axes (Figure 6-6). Along the second axis, sites receiving more frequent flows at high water levels were separated from those receiving high magnitude flows over shorter periods. This was taken as a facet of flow intermittency, describing the relative permanency of flows. Low flow intermittency scores on PC2 (i.e. high Ff and Wl) were observed particularly at sites with large, permanent wetlands (e.g. Algebuckina), consistent with modelled flow duration curves.

Stable water levels (PC3) and longer periods of inundation (PC4) were other distinguishing features associated with wetland size (i.e. wh.v). These last two facets describe subtle components of flow variation (together representing < 20 % of variation) that are nonetheless potentially important for water-dependent biota. They were decomposed by carefully analyzing interactions between water level, waterbody size and Tier 2 flow stage axes (Table 6-7, Figure 6-6): a negative relationship of all Tier 2 flow axes with water level along PC3 was indicative of water level stability, and a positive relationship of flow magnitude (Fm) and water level (Wl) along PC4 was considered evidence for time since inundation.

[17] See Figure 3-7 for clustering of all 29 flow metrics along the first two PCs. Abbreviations for metrics are defined in Table 3-3

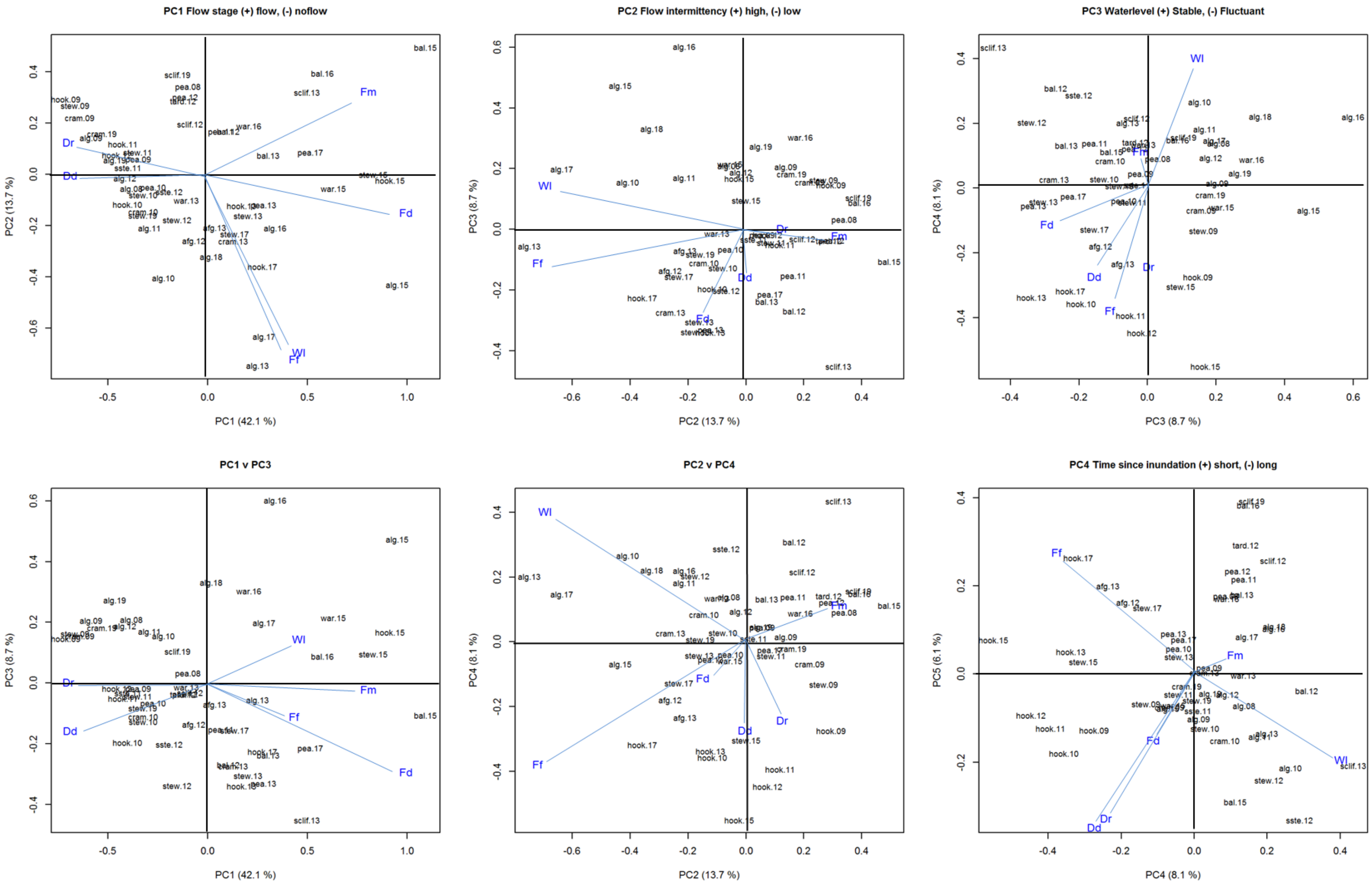


Figure 6-6: Decomposition of principal components into key facets of flow variability

Table 6-8: Results of the envfit analysis (fish mds 2)

Metric		NMDS1	NMDS2	r2	Pr(>r)	
days.q.spell	1	-0.87	-0.49	0.00	0.911	
qd.90	2	-0.81	-0.59	0.05	0.250	
qd.180	3	-0.74	-0.67	0.02	0.583	
qd.365	4	0.63	-0.78	0.06	0.239	
qd.max.yr	5	0.73	-0.68	0.12	0.035	*
zqd.max.yr	6	-0.59	0.81	0.12	0.050	*
zqd.spell	7	0.93	-0.36	0.01	0.749	
vol.q	8	-0.56	0.83	0.07	0.174	
qv.90	9	-0.22	-0.97	0.05	0.319	
qv.180	10	-0.17	-0.99	0.05	0.330	
qv.365	11	0.42	-0.91	0.06	0.199	
qve	12	0.31	-0.95	0.09	0.104	
qve.spell	13	0.60	-0.80	0.16	0.019	*
qve.st	14	0.21	-0.98	0.08	0.152	
qs.rlm.365	15	0.72	-0.69	0.02	0.597	
qs.fal.365	16	0.61	-0.79	0.07	0.156	
qs.rlm.max.yr	17	0.89	0.45	0.29	0.001	***
qs.fal.max.yr	18	0.87	-0.49	0.14	0.023	*
wl	19	-0.17	-0.98	0.15	0.018	*
wl.st	20	-0.02	-1.00	0.03	0.428	
zwld.low.spell	21	-0.64	0.77	0.13	0.035	*
wld.high.spell	22	-0.10	1.00	0.01	0.814	
zwld.low.365	23	-0.71	0.70	0.23	0.001	**
wld.high.365	24	0.72	-0.70	0.01	0.698	
qve.90	25	-1.00	0.03	0.08	0.135	
qve.365	26	-0.31	-0.95	0.06	0.234	
qve.730	27	-0.24	-0.97	0.10	0.076	.
wh.v	28	0.21	-0.98	0.23	0.002	**
fshap	29	0.98	0.20	0.01	0.719	

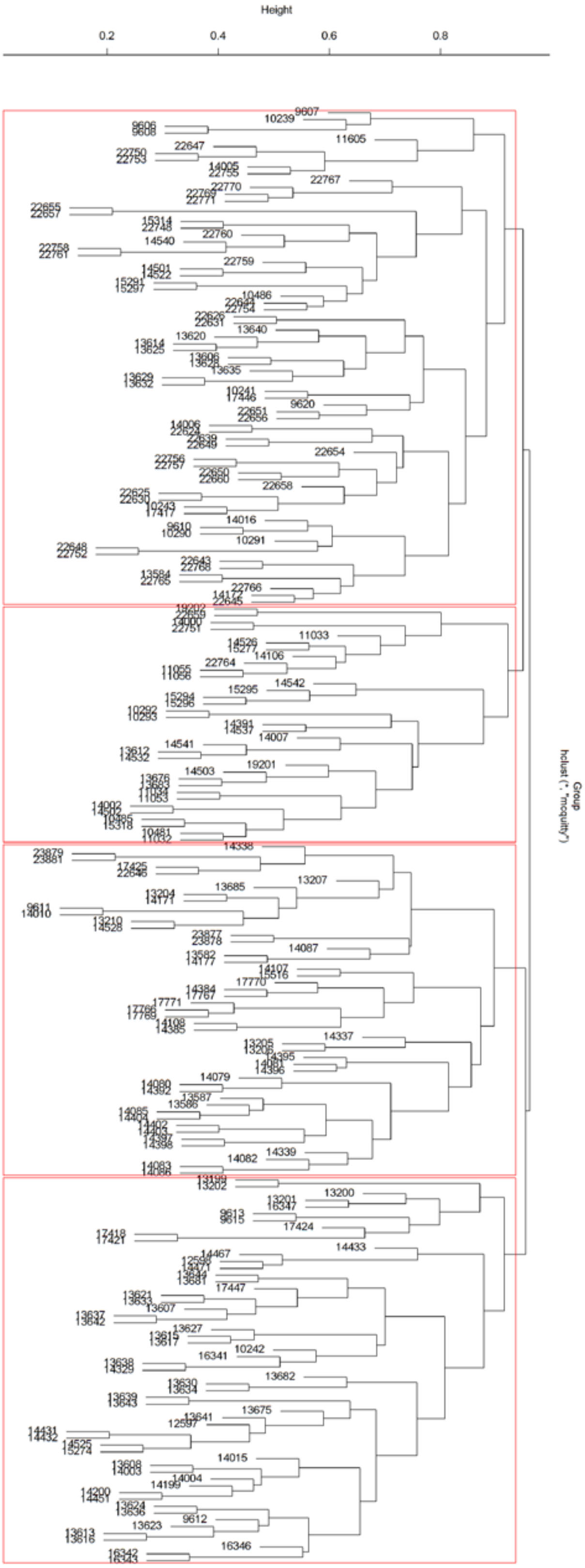
Significance: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
P values based on 2000 permutations.

PC axes	NMDS1	NMDS2	r2	Pr(>r)	
PC1	0.38	-0.92	0.08	0.121	
PC2	0.61	0.79	0.10	0.076	.
PC3	0.29	-0.96	0.20	0.006	**
PC4	0.88	-0.47	0.18	0.008	**
PC5	0.90	-0.44	0.17	0.015	*
PC6	-0.18	-0.98	0.01	0.727	
PC7	0.14	-0.99	0.01	0.849	
PC8	-0.70	-0.71	0.09	0.089	.
PC9	0.54	0.84	0.09	0.107	
PC10	1.00	0.00	0.06	0.222	

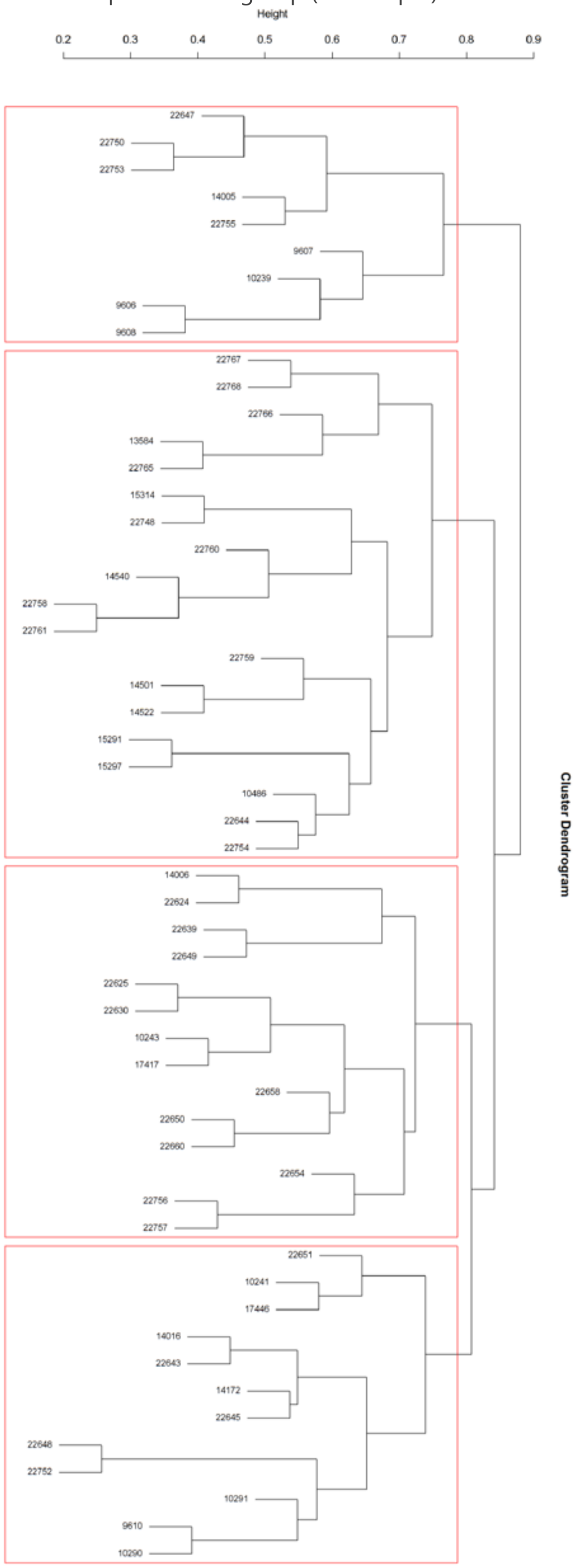
Significance: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
P values based on 2000 permutations.

D.3 Riparian and flood-dependent vegetation groups (labels are patch-ID)

Landscape-scale



Flood-dependent subgroup (of Group 1)



D.4 Floodplain assemblages and options for extent of inundation modelling

The following tables/figures present additional data as part of the riparian and floodplain vegetation analysis (Section 3.4.5).

- Group 1 includes riparian woodlands of the inner floodplains and floodouts of the western rivers, distinguished by several dominant riparian and floodplain shrub and tree species along the rivers and near waterholes (*Eucalyptus coolabah, E. camaldulensis, Acacia cambagei, A. salicina, A. stenophylla, Duma florulenta*) as well as flood-responsive understorey species. The group occurs widely across the western rivers in close proximity to active drainage lines, floodplains and swamps. It corresponds well with the drainage line, floodplain and swamp delineation of Brandle (1998) (e.g. see Landform Elements in Figure 2-10), shown by the correspondence of Group 1 with numbered 1s in the Landform Element (Figure 6-7).
- Group 2 includes grasslands (*Enneapogon avenaceus, Eragrostis dielsii*), salt-tolerant chenopod shrublands (*Nitraria billardieri*) as well as more widespread species that respond to wet conditions (*Acacia ligulata, A. victoriae*). The group occurs around lake margins or saline sandy reaches, particularly in mid- to lower reaches of the western rivers.
- Group 3 includes sparse woodlands and low shrublands of the outer floodplains and rocky headwaters, distinguished by several widespread species of the stony plains (e.g. *Acacia aneura, A. tetragonophylla*) (Brandle, 1998).
- Group 4 includes low shrublands (*Sclerolaena intricata, Maireana aphylla*) of stony gilgais and claypans in localized areas of the western rivers, and include some important fodder species (e.g. *Atriplex nummularia*).

Table 6-9: Results of the envfit analysis

Landscape GIS var	NMDS1	NMDS2	r2	Pr(>r)	Sig
Elev	-0.29272	0.956	0.065	0.0024988	**
DistWc	0.53806	0.843	0.068	0.0009995	***
ElevWc	-0.302	0.953	0.063	0.0029985	**
DistLak	-0.11291	0.994	0.014	0.2118941	ns
ElevLak	-0.36901	0.929	0.018	0.1429285	ns
ElevCh	0.3797	0.925	0.142	0.0004998	***
DistWH	0.46307	0.886	0.124	0.0004998	***
LFZ	0.37171	0.928	0.410	0.0004998	***

Significance: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
P values based on 2000 permutations

Patch scale var	NMDS1	NMDS2	r2	Pr(>r)	Sig
Site.Slope	0.60102	0.799	0.043	0.065967	ns
Strew.size	0.89682	0.442	0.057	0.0174913	*
Strew.cov	0.97254	0.233	0.227	0.0004998	***
Lit.cov	-0.81116	-0.585	0.110	0.0004998	***
Perc.Cl	0.48311	-0.876	0.282	0.0004998	***

Significance: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
P values based on 2000 permutations
80 observations deleted due to missing data

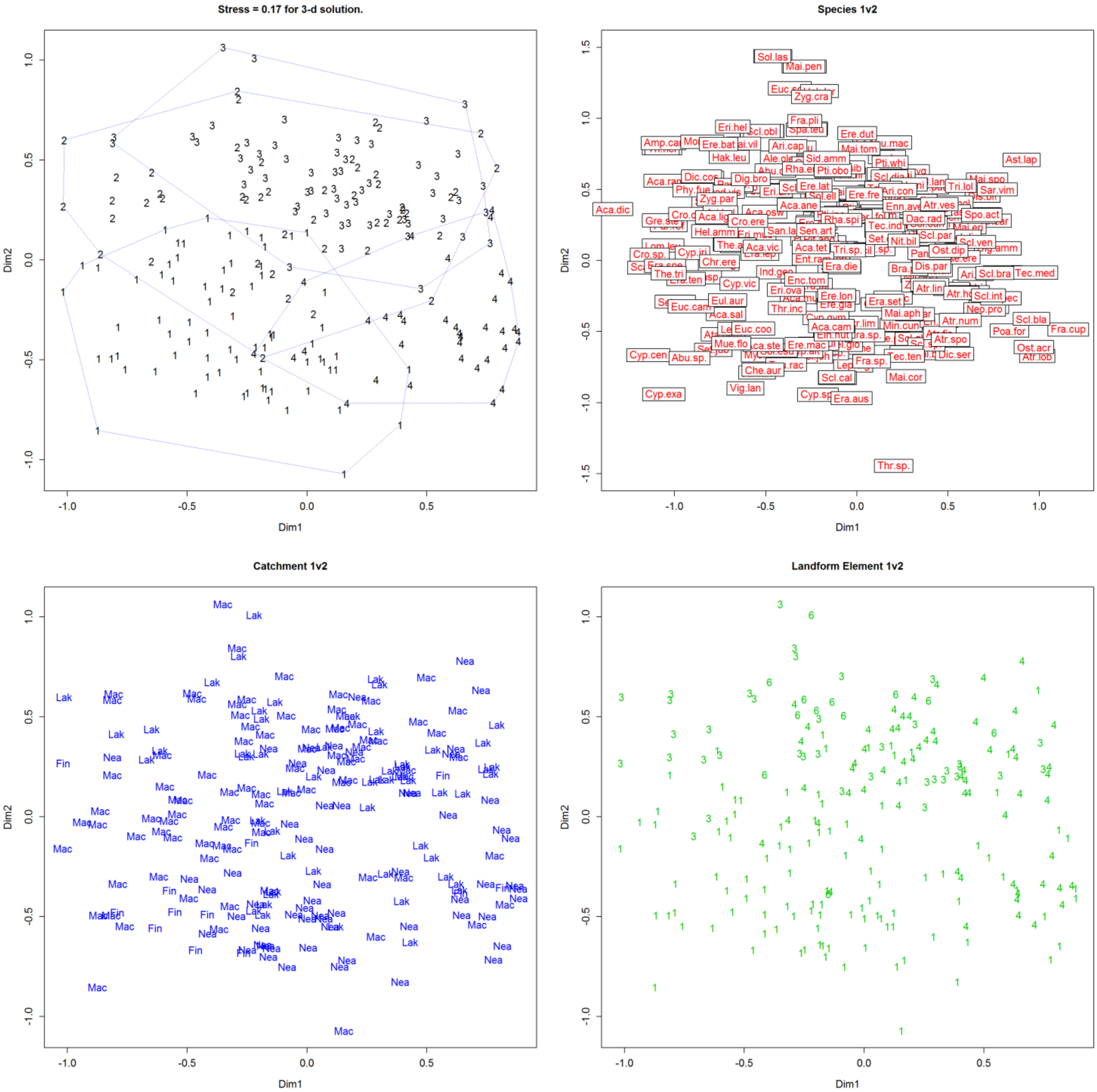


Figure 6-7: Structure of floodplain and flood-dependent assemblage groups (GIS cut-off)

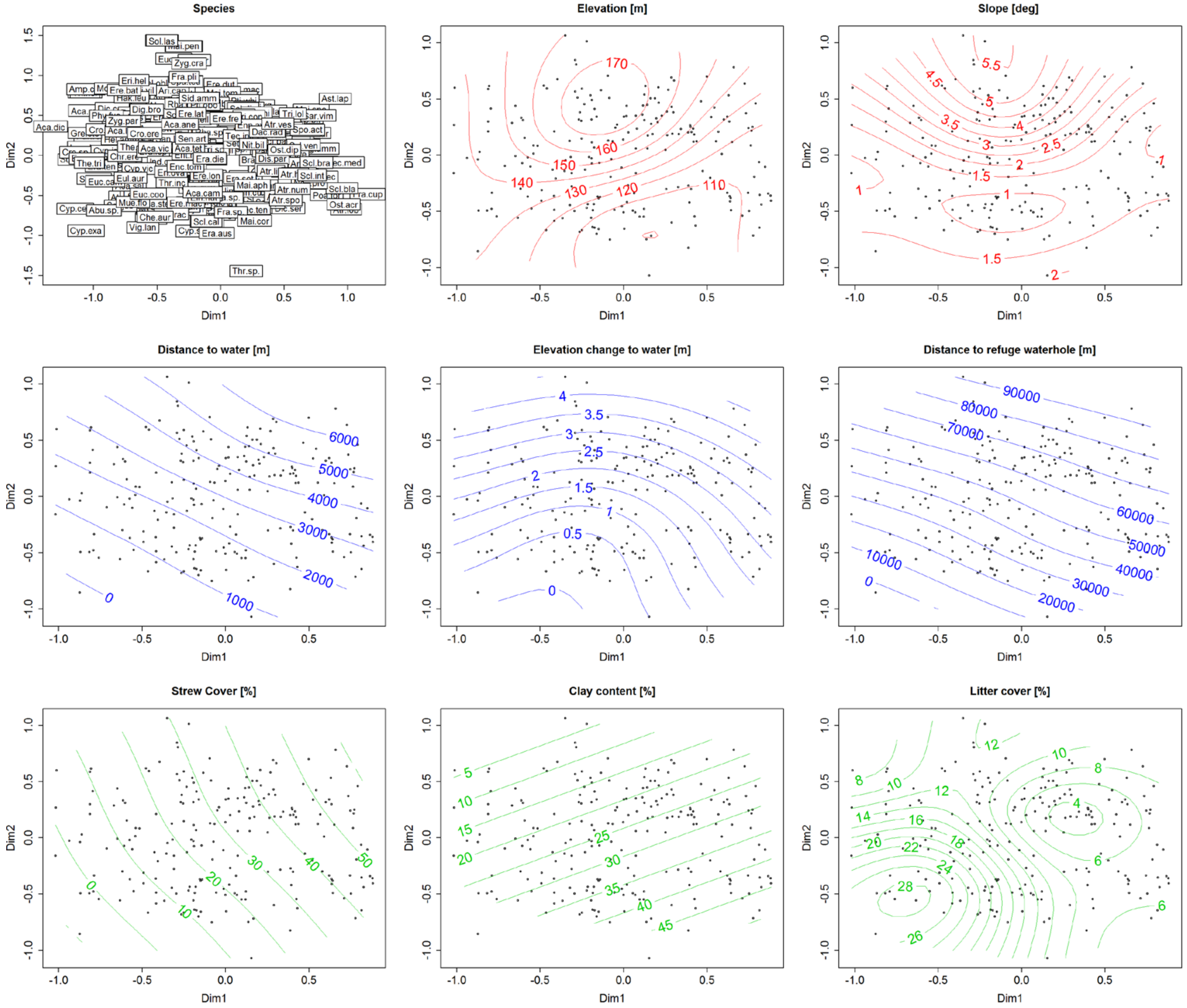


Figure 6-8: Environmental surfaces, modelled for four vegetation groups (cf. Figure 6-7)

Table 6-10: Indicator species four vegetation flood-groups analysed based on GIS data cutoffs

Species	Spp.Abb	common name	indval†	Habitat preference‡
Group 1. Inner floodplain group (riparian woodland)				
Acacia cambagei	Aca.cam	Gidgee	0.477	above wl clay soils
Acacia salicina	Aca.sal	Cooba	0.124	above wl
Acacia stenophylla	Aca.ste	Eumong, River Cooba	0.147	at or above wl
Chenopodium auricomum	Che.aur	Golden Goosefoot	0.416	above wl, clay soils
Cyperus alterniflorus	Cyp.alt		0.055	at or above wl
Cyperus exaltatus	Cyp.exa	Tall Flat Sedge	0.055	at wl
Cyperus victoriensis	Cyp.vic	Yelka	0.177	in channel or on banks
Einadia nutans	Ein.nut	Climbing Saltbush	0.335	heavy soils
Enchylaena tomentosa	Enc.tom	Ruby Saltbush	0.338	variable, widespread
Eragrostis australasica	Era.aus	Canegrass	0.126	intermittent flooding clay soils
Eragrostis setifolia	Era.set	Neverfail[1]*	0.228	variable, periodically flooded
Eriachne ovata	Eri.ova		0.239	variable, rocky and outwash plain
Eucalyptus camaldulensis	Euc.cam	River Redgum	0.215	at or above wl
Eucalyptus coolabah	Euc.coo	Coolabah	0.790	above wl
Eulalia aurea	Eul.aur	Silky Browntop	0.415	variable, usually moist areas, w rg
Muehlenbeckia[18] *florulenta*	Mue.flo	Lignum	0.313	at wl
Sclerolaena bicornis	Scl.bic	Goathead Burr[3]	0.202	variable, alluvial flats
Teucrium racemosum	Teu.rac	Grey Germander	0.137	infrequent major flooding
Group 2. Lacustrine group (saline sandy flats or salt lake margins)				
Acacia ligulata	Aca.lig	Sandhill Wattle	0.350	lake edges, sandy soils
Acacia victoriae	Aca.vic	Prickly Wattle[2]	0.092	variable, widespread, wet season increaser
Atriplex holocarpa	Atr.hol	Pop Saltbush[3]	0.194	variable, short-term flood responder
Atriplex lindleyi	Atr.lin	Flat-Top Saltbush	0.108	depressions or low-lying areas
Crotalaria eremaea	Cro.ere	Loose-Flowered Rattlepod	0.392	sandy soils w mulga, ephemeral creeks
Cyperus gymnocaulos	Cyp.gym	Spiny Flatsedge	0.070	at or above wl
Enneapogon avenaceus	Enn.ave	Common Bottlewashers[1]	0.364	calcareous sandy soils, wet season increaser
Eragrostis dielsii	Era.die	Mulka[1]	0.369	calcareous sandy soils, short-lived perennial
Maireana ciliata	Mai.cil	Hairy Fissue-Weed	0.082	calcareous soils
Nitraria billardierei	Nit.bil	Nitre-Bush[3]	0.278	above wl, saline creek flat
Sclerolaena diacantha	Scl.dia	Grey Copperburr[2]	0.241	variable, widespread
Sida ammophila	Sid.amm	Sand Sida	0.222	variable, widespread forb
Tecticornia indica	Tec.ind		0.153	
Triraphis mollis	Tri.mol	Purple Plume Grass[3]	0.257	widespread sandy soils
Zygochloa paradoxa	Zyg.par	Sandhill Canegrass[3]	0.398	sandy soils and dunes, imp for dune stabilization
Group 3. Outer floodplain group (stony soils on outer margin of floodplain)				
Acacia aneura	Aca.ane	Mulga	0.369	red sands and gravels
Acacia tetragonophylla	Aca.tet	Dead Finish[2]	0.289	red sands and gravels
Aristida contorta	Ari.con	Kerosene Grass[2]	0.273	variable, widespread
Atriplex vesicaria	Atr.ves	Bladder Saltbush[1]	0.507	variable, widespread, alluvial plains w Mitchell grass
Eremophila freelingii	Ere.fre	Rock-Fuchsia Bush[3]	0.474	gibber soils, outer floodplain
Eremophila latrobei	Ere.lat	Crimson Turkey-Bush[3]	0.294	stony soils
Eremophila serrulata	Ere.ser		0.076	
Maireana georgei	Mai.geo	Satiny Bluebush[1]	0.196	red sand, w Mulga
Monachather paradoxus	Mon.par	Bandicoot Grass[1]	0.067	sandy or gravelly soils
Sclerolaena cuneata	Scl.cun		0.217	
Sclerolaena lanicuspis	Scl.lan		0.082	
Sclerolaena longicuspis	Scl.lon	Long-Spined Poverty-Bush[3]	0.225	gibber soils
Sclerolaena uniflora	Scl.uni		0.241	
Senna artemisioides	Sen.art	Silver Cassia	0.324	variable, widespread
Solanum ellipticum	Sol.ell	Velvet Potato-Bush	0.233	variable, widespread, mostly sandy soils
Tripogon loliiformis	Tri.lol	Five-Minute Grass	0.378	gibber soils, rain responder resurrection plant
Group 4. Claypans or gilgai group (clay soils)				
Aristida anthoxanthoides	Ari.ant	Pale Wiregrass	0.071	periodic flooding, w Mitchell grass

[18] *Now* Duma

Astrebla pectinata	Ast.pec	Barley Mitchell Grass[1]	0.317	gilgai or alluvial sands, wet season increaser
Atriplex nummularia	Atr.num	Oldman Saltbush[2]	0.575	variable, widespread, clay soils low-lying areas
Atriplex spongiosa	Atr.spo	Pop Saltbush[3]	0.301	variable, short-term flood responder
Eragrostis parviflora	Era.par	Weeping Lovegrass	0.098	moist clay soils, near permanent water
Maireana aphylla	Mai.aph	Cottonbush	0.317	variable, alluvial flats w Bladder Saltbush
Sclerolaena blackiana	Scl.bla	Black's Copperburr	0.141	clay soils, uncommon species
Sclerolaena intricata	Scl.int	Tangled Poverty-Bush[3]	0.403	clay depressions and gilgai
Tecticornia medullosa	Tec.med	None Recorded	0.097	
Group S. Desert springs group				
Cyperus laevigatus	Cyp.lae	Bore Drain Sedge[2]	0.970	below wl, grows in shallow waters of artesian bores
Phragmites australis	Phr.aus	Common Reed[2]	0.177	at wl, shallow intermittent floods or marshes

[1]fodder species, [2]drought fodder species (partially grazed), [3]unsuitable for stock (overgrazing increasers)

† Relative contribution to group structure based on indicator species analysis (Roberts 2013)

‡ Notes from (Cunningham et al. 2011)

D.5 Extent of inundation time-series model

Landsat imagery was processed for areal extent of inundation on each of the eight scene dates (see Table 3-1); this was a dependent variable using selected flow metrics (indicators of the axes of flow variability: Appendix E) as predictor variables in a Generalized Linear Modelling approach (Figure 6-9). The relationships were then used to predict time-series of inundation extents from 1999–2013. The distributions of indicator species and species groups were compared to the results of models to validate them against understanding of flood regimes supporting growth and water requirements (Roberts and Marston 2011).

Floodplain inundation extents estimated from satellite imagery (Appendix D) were modelled against Tier 1 flow indicators: qv.90, qd.90 and zqd.spell. The results of the four reaches combined were fit to a hockey-stick model for modelling time-series which could be used to model floodplain extents under natural and altered mining scenarios. The model was insufficient for predicting a flood inundation extent (based on the eight scene dates), but shows promise. The model outputs are presented below as a ‘proof of concept’ for future modelling using a more comprehensive NDVI time series.

The hockey stick model (Figure 6-9) is described using qd.90 as the sole predictor, X (Eqn 1):

=IF(X<=30,0.09195,0.00000762*(X^3) - 0.00064*(X^2) + 0.0132*X + 0.0736) Eqn 1

Table 6-11 Results of model term fit

Coefficients:	Estimate	SE	*t*	P
(Intercept)	7.36E-02	3.81E-02	1.931	0.06444
qd.90	1.32E-02	6.40E-03	2.055	0.0501
I(x1^2)	-6.40E-04	2.32E-04	-2.759	0.01048
I(x1^3)	7.62E-06	2.10E-06	3.634	0.00121

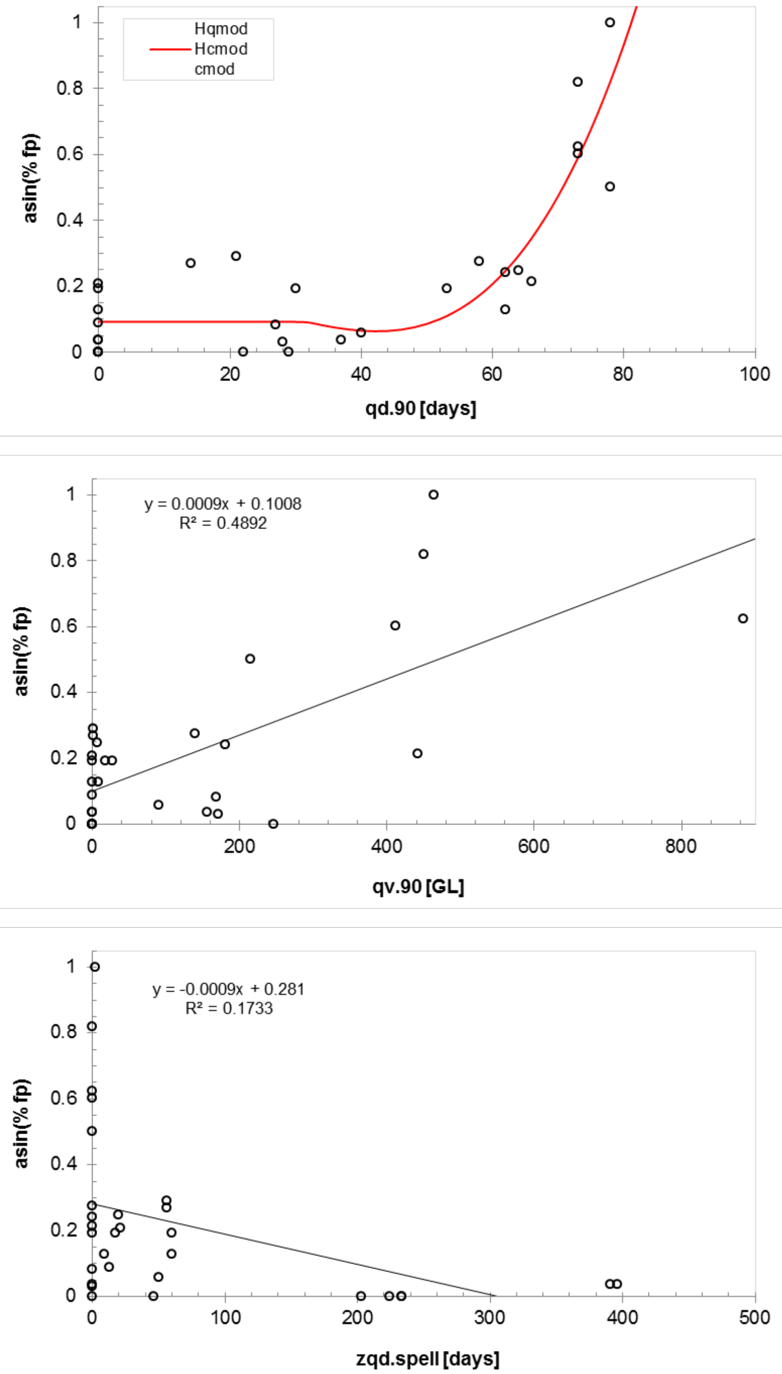


Figure 6-9: Models predicting floodplain inundation (derived from satellite image analysis) as a function of 90-d antecedent flow (duration: qd.90 and volume: qv.90) and length of dry spell (zqd.spell)[19]

[19] All models were significant at 95%, although qd.90 is considered the only viable predictor for a floodplain time-series model.

E. Modelling a non-standard floodplain

The term 'non-standard floodplain' is used to describe the mosaic-like distribution of vegetation in the floodplains of the central arid zone rivers (Section 3.4.5).

Subtle shifts in relative dominance of species on the western rivers floodplains across the microrelief need explanation, both for management of conservation values and to better understand how hydrology interacts with other drivers in assembling floodplain communities. An approach was trialled in this study using modelled stage height (Montazeri and Osti 2014) and floodplain elevation cross-sections. Due to the unreliability of the modelled ARI for stage heights above cease-to-flow (Montazeri and Osti 2014), the outputs of the analysis are considered preliminary, but the approach could be useful to identify the drivers of floodplain assemblages. The results are included here to demonstrate the method.

Following Montazeri and Osti (2014), water-level exceedance probabilities (Figure 6-10) were set up as hypothetical flow bands corresponding to the Annual Recurrence Interval (ARI) estimates of 1:2, 1:5, 1:10, 1:20, 1:50 and 1:100 year events (Table 6-12). These estimates were calculated from all the events exceeding cease-to-flow water levels, modelled from 1950–2013 (although, as noted, estimates of stage height above cease-to-flow have not been accurately modelled). Flow bands were then overlaid on surveyed cross-sections of the floodplain (Figure 6-12 to Figure 6-15). Stage-discharge curves were also estimated for locations based on 2-dimensional modelling of the cross-sections (Figure 6-11). The modelling of ARIs suggests that the four reaches receive floods at different magnitudes (i.e. flow bands) and return times (Figure 6-12 to Figure 6-15).

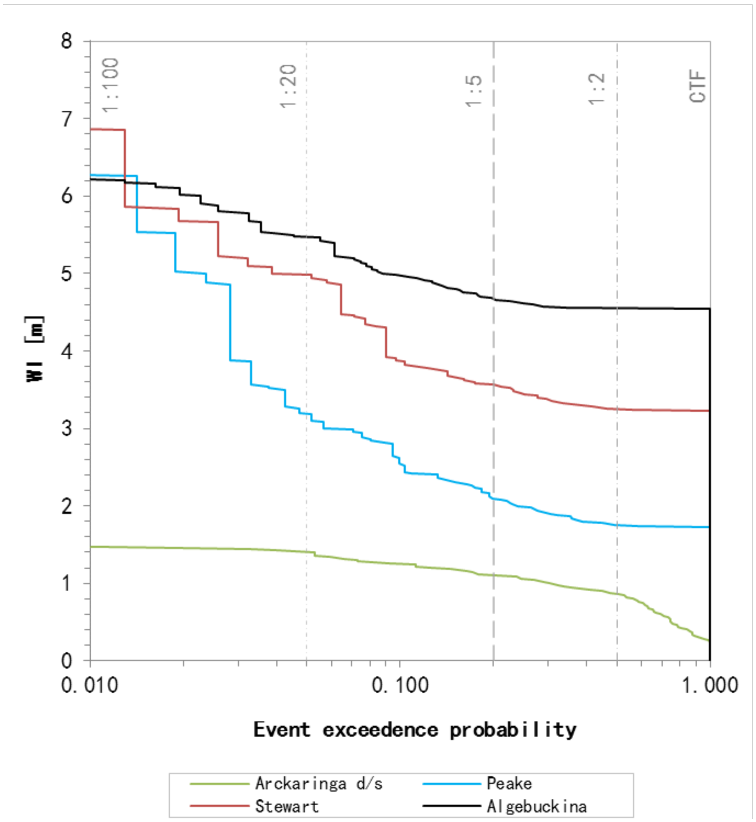


Figure 6-10: Probability of exceeding cease-to-flow levels for reaches based on modelled stage height

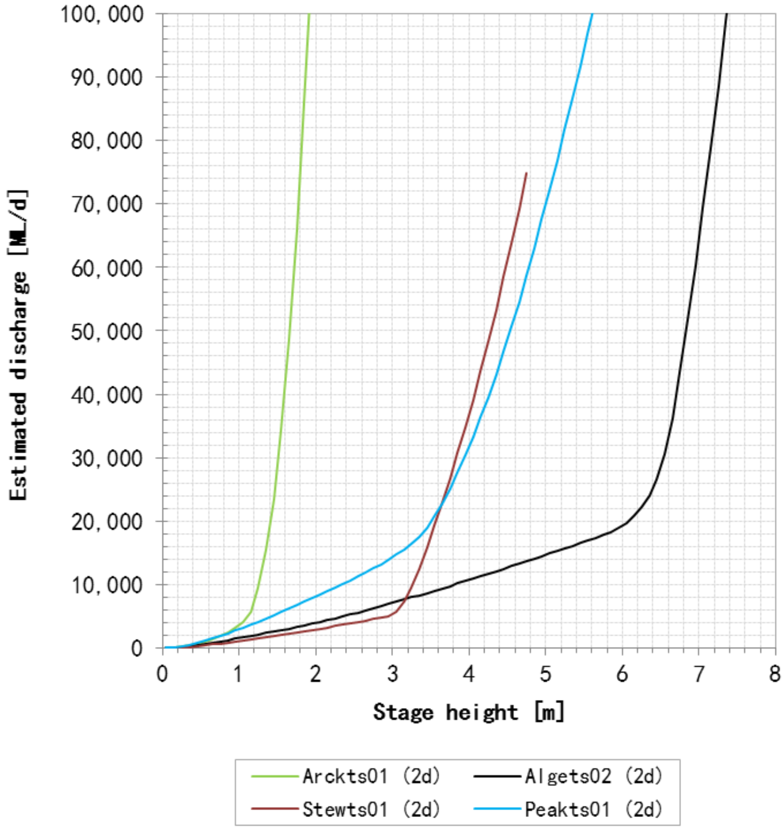


Figure 6-11: Estimated stage-discharge curves for floodplains, Neales–Peake catchment

Table 6-12: Modelled ARIs and flow bands for sites on the Neales and Peake rivers. Note that modelled stage heights above cease-to-flow level are unreliable (Montazeri and Osti 2014)

Event	Upper Peake		Mid Peake		Lower Peake		Upper Neales				Mid Neales		Lower Neales	
	Arckaringa[1]		*Peake*[5]		*Warrawaroona*[5]		*Stewart*[2]		*South Stewart*[2]		*Algebuckina*[3]		*South Cliff*[4]	
	*Flow band**	Wl_{Pred}†	*Flow band*	Wl_{Pred}	*Flow band*	Wl_{Pred}	*Flow band*	Wl_{Pred}	*Flow band*	Wl_{Pred}	*Flow band*	Wl_{Pred}	*Flow band*	Wl_{Pred}
1:2	inch	0.65	inch	0.03	inch	0.23	inch	0.02	inch	0.17	inch	0.06	inch	0.12
1:5	ob	0.85		0.38		0.53		0.32		0.37		0.21		0.22
1:10		1.00		0.88		1.13	ob	0.57		0.77		0.46		0.32
1:20	fl	1.15		1.48	ob	2.33	fl	1.77	ob	1.07		0.96	ob	0.62
1:50			ob	3.28	fl	4.03		2.47	fl	2.17			fl	1.12
1:100		1.25	fl	4.58				3.67				1.66		1.42

Superscripts refer to site locations

*inch=in-channel, ob=overbank, fl=flood

†tabulated as relative water level (height above cease-to-flow)

Site	W_{fp}	CTF	BF	$Diff_{BF-CTF}$
Arckaringa	2400	0.3	0.8	0.5
Stewart	639	3.2	3.7	0.5
South Stewart	950	2.5	3.9	1.4
Algebuckina	1,000	4.5	7.9	3.4
Peake Crossing	303	1.7	4.7	3.0
South Cliff	1600	2.4	3.0	0.6
Warrawaroona	800	2.4	4.7	2.3

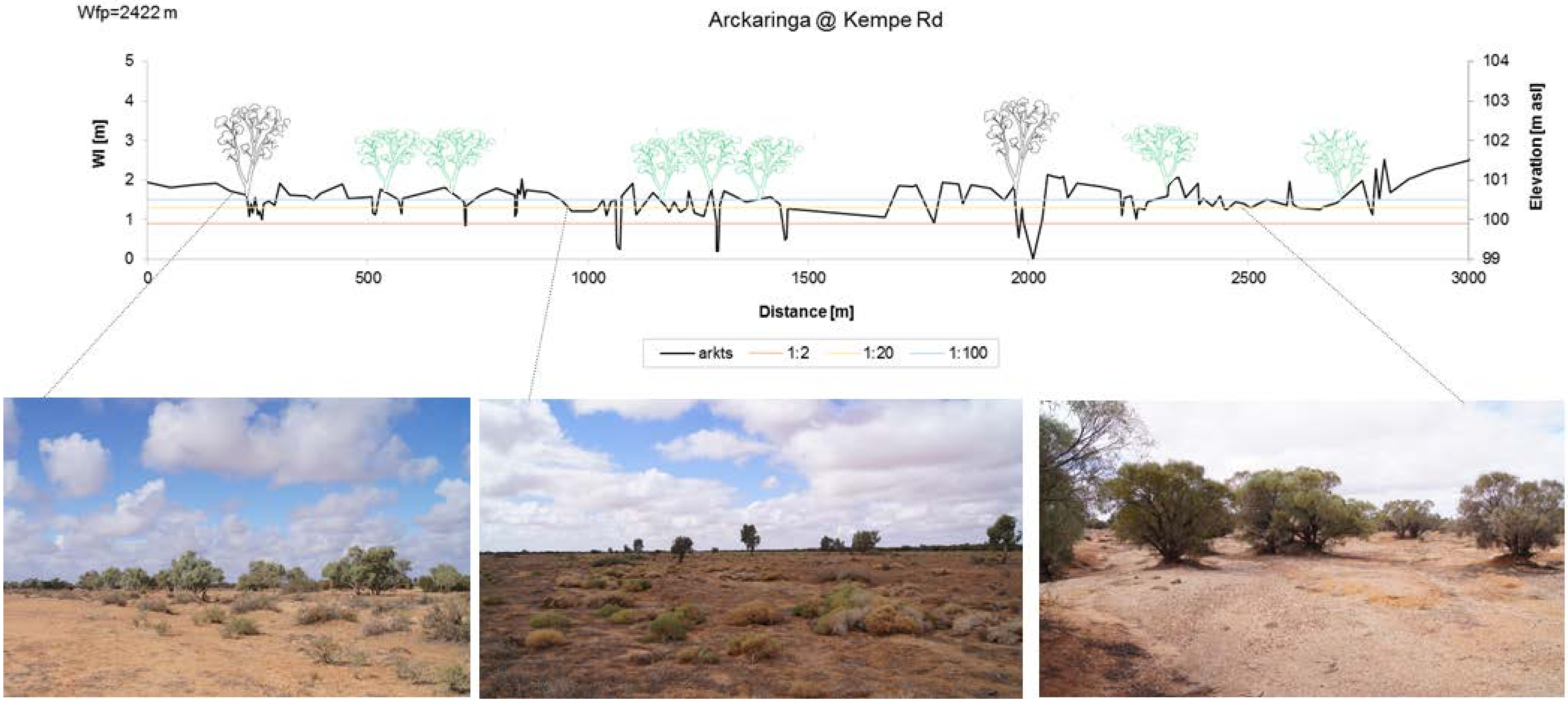


Figure 6-12: Floodplain cross-sections of Arckaringa Creek, indicating relationships between microrelief, flood regime and species occurrence

Black trees represent Coolabah (*Eucalyptus coolabah*); green trees represent Gidgee (*Acacia cambagei*). Flood bands (ARI) are depicted as horizontal lines (orange=1:2 or 1:5 events, yellow=1:10 or 1:20, blue=1:50 or 1:100). Flood ARIs are based on unreliable modelled data. Photos display vegetation along the cross-sections.

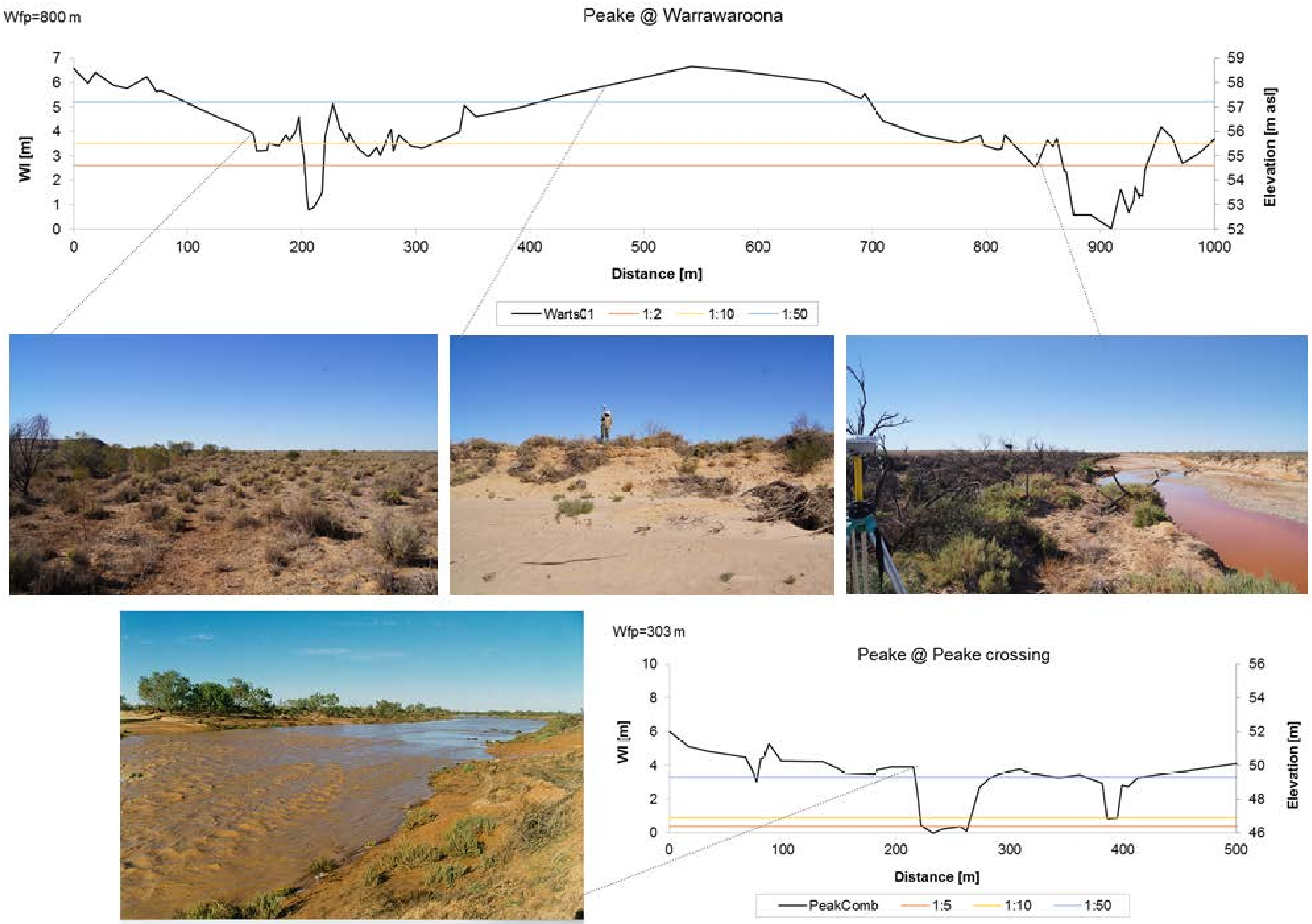


Figure 6-13: Floodplain cross-sections of Peake River, showing relationships between microrelief, flood regime and species

Black trees represent Coolabah (*Eucalyptus coolabah*); green trees represent Gidgee (*Acacia cambagei*). Flood bands (ARI) are depicted as horizontal lines (orange=1:2 or 1:5 events, yellow=1:10 or 1:20, blue=1:50 or 1:100). Flood ARIs are based on unreliable modelled data. Photos display vegetation along the cross-sections.

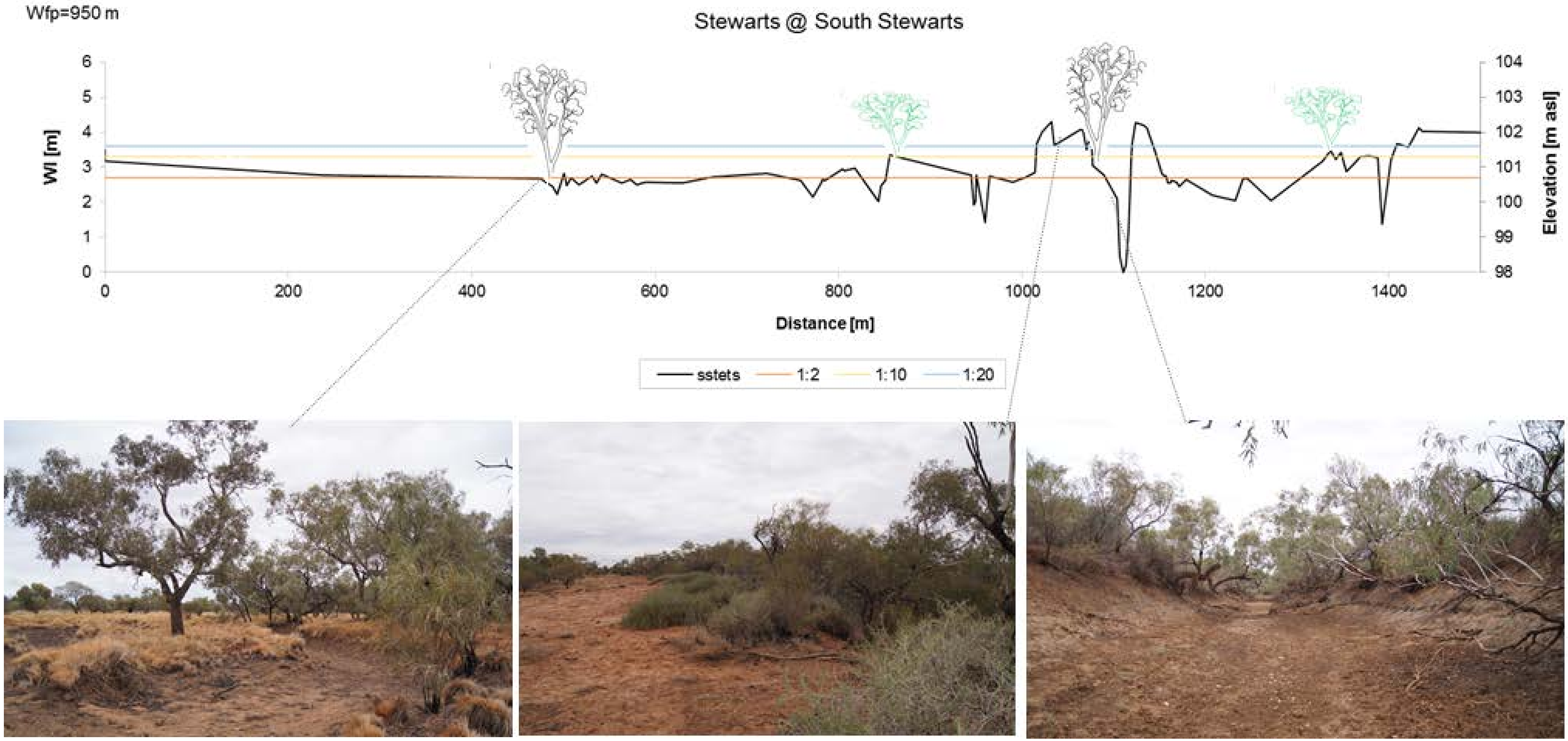


Figure 6-14: Floodplain cross-sections of Neales River at South Stewarts Waterhole, showing relationships between microrelief, flood regime and species

Black trees represent Coolabah (*Eucalyptus coolabah*); green trees represent Gidgee (*Acacia cambagei*). Flood bands (ARI) are depicted as horizontal lines (orange=1:2 or 1:5 events, yellow=1:10 or 1:20, blue=1:50 or 1:100). Flood ARIs are based on unreliable modelled data. Photos display vegetation along the cross-sections.

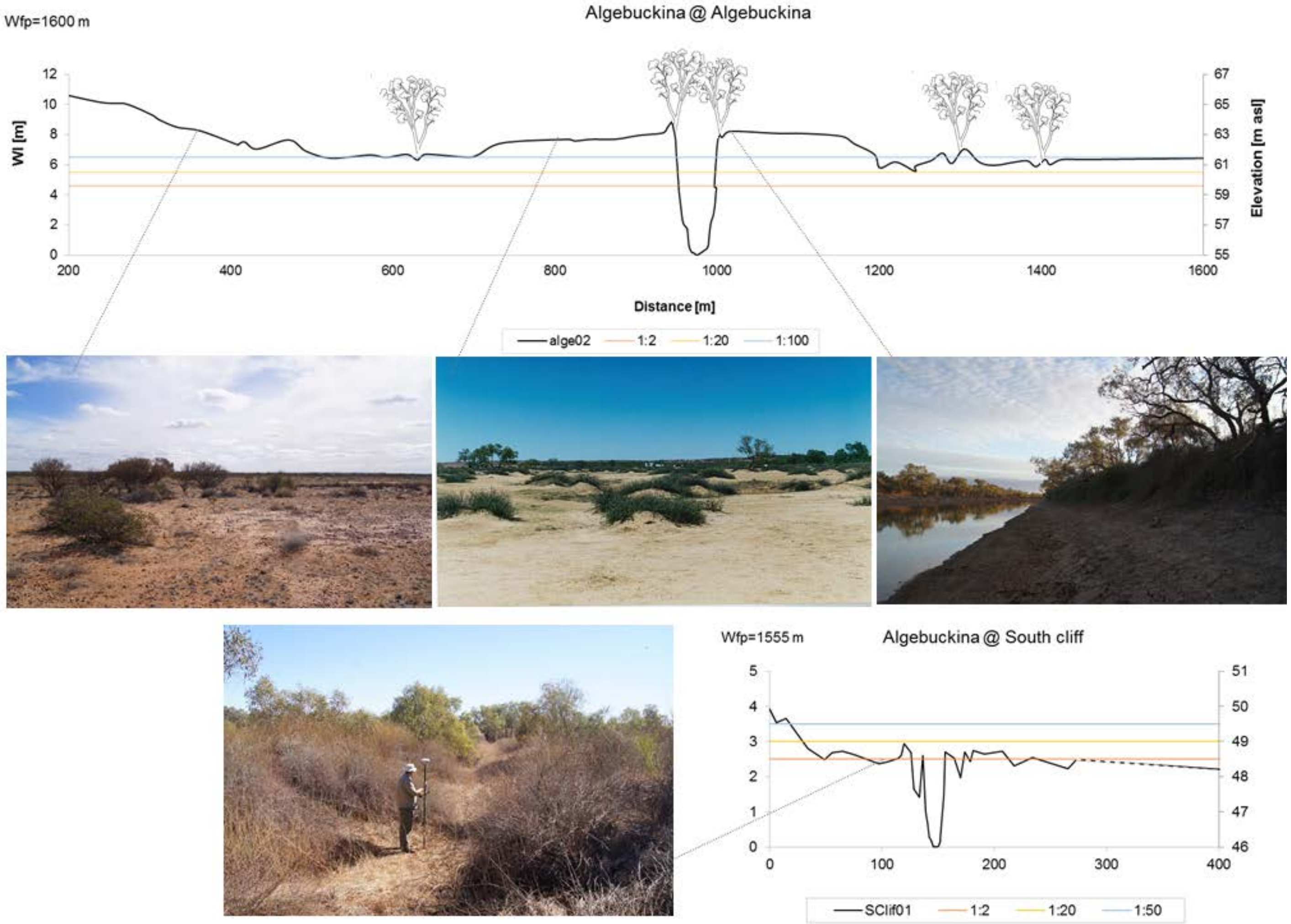


Figure 6-15: Floodplain cross-section of Neales River at Algebuckina, showing relationships between microrelief, flood regime and species

Black trees represent Coolabah (*Eucalyptus coolabah*); green trees represent Gidgee (*Acacia cambagei*). Flood bands (ARI) are depicted as horizontal lines (orange=1:2 or 1:5 events, yellow=1:10 or 1:20, blue=1:50 or 1:100). Flood ARIs are based on unreliable modelled data. Photos display vegetation along the cross-sections.

7. Units of measurement

Name of unit	Symbol	In terms of other metric units	Quantity
day	d	24 h	time interval
gigalitre	GL	10^6 m^3	volume
gram	g	10^{-3} kg	mass
hectare	ha	10^4 m^2	area
hour	h	60 min	time interval
kilogram	kg	base unit	mass
kilolitre	kL	1 m^3	volume
kilometre	km	10^3 m	length
litre	L	10^{-3} m^3	volume
megalitre	ML	10^3 m^3	volume
metre	m	base unit	length
microgram	µg	10^{-6} g	mass
microliter	µL	10^{-9} m^3	volume
milligram	mg	10^{-3} g	mass
millilitre	mL	10^{-6} m^3	volume
millimetre	mm	10^{-3} m	length
minute	min	60 s	time interval
second	s	base unit	time interval
tonne	t	1000 kg	mass
year	y	365 or 366 days	time interval

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