TECHNICAL REPORT

PRELIMINARY STUDY OF THE INTERACTION BETWEEN LAND USE AND PERCHED WETLAND HYDRO-ECOLOGY ON THE SOUTHERN FLEURIEU PENINSULA

2012/19

Government of South Australia
Department for Water

PRELIMINARY STUDY OF THE INTERACTION BETWEEN LAND USE AND PERCHED WETLAND HYDRO-ECOLOGY ON THE SOUTHERN FLEURIEU PENINSULA

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Science, Monitoring and Information Division
Department for Water

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FOREWORD

South Australia's Department for Water leads the management of our most valuable resource—water.

Water is fundamental to our health, our way of life and our environment. It underpins growth in population and our economy—and these are critical to South Australia's future prosperity.

High quality science and monitoring of our State's natural water resources is central to the work that we do. This will ensure we have a better understanding of our surface and groundwater resources so that there is sustainable allocation of water between communities, industry and the environment.

Department for Water scientific and technical staff continue to expand their knowledge of our water resources through undertaking investigations, technical reviews and resource modelling.

Allan Holmes

CHIEF EXECUTIVE

DEPARTMENT FOR WATER

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SUMMARY

Background and Scope

This report summarises the findings of a three-year study on the interaction between the land use, hydrology and ecology of perched wetland systems. The study was conducted on the southern Fleurieu Peninsula in south-central South Australia on wetlands associated with first and second order streams. Higher rainfall areas in the region have a mean annual rainfall in excess of 900 mm and support a large number of perched wetland systems of national conservation significance.

The investigation was intended to develop a preliminary understanding of the hydrology and ecology to inform the additional work required to underpin sustainable management of perched wetlands from a water resource management perspective. The questions examined for perched wetlands were informed by the potential management responses:

- What proportional level of forest development creates an impact on catchment hydrology that would potentially change wetland ecological character?
- Are buffers an effective means with which to manage hydrological impacts on wetlands resulting from plantation forestry?

Three perched wetlands located on second order streams and of comparable catchment area (88–151 ha), but under contrasting dominant land uses (pine plantation, native woodland and perennial pasture), were instrumented for hydrological and volumetric soil moisture data collection. Conceptual understanding of wetland hydrology and plant ecology was developed through analysis of monitoring data, soil physical description/hydraulic characterisation and vegetation survey. Hydrological and ecological models were constructed based on this understanding, but incorporated considerable simplification of the complex water movement processes identified within catchments and wetlands.

The essence of the project was a modelling study to assess the changes to surface water runoff and wetland ecology for a hypothetical 100 ha wetland-catchment system under various land use compositions. Modelling scenarios manipulated the proportional area of planting within the catchment (planting fraction) and the width of buffer zones. Planting fraction and buffer width scenarios modelled were selected to bracket state policy guidelines and incorporate existing plantation development levels. Surface water runoff and predicted vegetation community were assessed under each scenario.

Results and findings

Study catchments were comparable in area, topography and in soil type but wetland hydrology at the three field sites differed markedly. Watertables within wetlands in the plantation and native forest catchments were seasonal—storages at one site dried completely, while at the other watertables declined to more than a metre below the surface. In contrast permanent surface saturation and baseflow were observed at the pasture site. These were supported by a local confined groundwater flow system, which constitutes a previously unknown water movement pathway for perched wetlands in the region.

The relative importance of overland flow, throughflow via A and B horizons and the presence of any confined groundwater system, exerted a strong influence on the observed wetland hydrology and ecology. Land use and water resource development will affect each of these in different ways. In future effective modelling, monitoring and management of perched systems in the region will need to be based on an understanding of each of these processes, interactions between these and the manner in which land use affects this.

Modelling focused on hydrological processes as observed at the two seasonal wetlands. Results indicated all other land use combinations, planting fractions and buffer widths produced less runoff than pasture. Runoff reduction compared with the modelled pasture response was found to follow an inverse linear relationship with the percentage forested area in the scenario.

Statistical models of vegetation, at functional group level, support field observations that semi-aquatic plant functional groups exhibit the best physiological adaptation to the seasonal hydrology of perched wetlands. True aquatic species will generally be restricted to the wettest areas of these swamps, but will be common in perennial perched systems. Evidence suggests that wetland plants with a woody growth habitat will persist longer within a water limited wetland than other amphibious groups.

The effects of buffers on wetland hydrology and ecology were dependent upon planting fraction. At low planting proportions, buffers were not greatly influential but increasing buffer area in 100% forested modelling scenarios is predicted to have a major effect on the observed vegetation community.

Modelling suggests a 100 ha catchment comprising 30% pine plantation and a 20 metre buffer over a 37 year forest rotation would reduce the median annual streamflow expected under perennial pasture with no water resource development by 21%. The impacts of development at this level are not considered likely to result in any major change in perched wetland ecological character, though some shift in the relative abundance of functional groups (and therefore species) may occur over time.

To fully understand wetland hydro-ecological variation the relative importance of soil water movement processes described above, as well as the land use, require consideration. The relative size of the wetland and the contributing catchment is also an important determinant of hydrology that was not examined in this study and remains a knowledge gap.

Stream flow data are of limited duration for the study region and was collected at a scale inconsistent with that used for current water allocation planning. As a result, this presents limited opportunity for analysis at this stage. Modelled responses remain unverified over time and across the range of spatial variability present in perched wetland catchments. The preliminary result presented should be seen as indicative of regional patterns until a longer dataset is acquired to enable more robust interpretation.

The project has improved understanding of perched wetland hydrological and ecological processes, in particular the pathways by which water moves to perched wetlands and the likely plant functional groups perched wetlands will support.

Modelling undertaken was essentially a trial of the methods employed and additional research is required to improve certainties around both wetland and catchment hydrology and wetland plant responses to improve water resource management.

1. INTRODUCTION

1.1. PROJECT CONTEXT

Fleurieu Peninsula wetlands are recognised as important ecological assets and the *Swamps of the Fleurieu Peninsula* are a listed Threatened Ecological Community under the *Environment Protection and Biodiversity Conservation Act (1999)*, with a status of critically endangered. Many of the threatening processes identified for Fleurieu Peninsula Swamps and wetlands relate directly to alteration of their hydrology (Duffield and Milne 2000, Harding 2005).

This report presents the findings of a joint National Water Initiative – Government of South Australia funded investigation conducted by the Department for Water (previously Department of Water, Land and Biodiversity Conservation) between June 2007 and May 2010. A major motivation for the study was the lack of quantitative understanding of the hydrology and ecology of small perched wetlands in the southern Fleurieu Peninsula region and the relative sensitivity of these systems to land use change due to their reliance on local processes (Barnett and Rix 2006).

The findings of this project are expected to contribute to future water policy development for the Mount Lofty Ranges in South Australia.

1.2. SCOPE OF INVESTIGATION

This project was primarily a modelling study to assess changes to wetland hydrology and ecology under various forest planting scenarios. Test cases for each of the modelling scenarios were chosen based on policy scenarios consistent with the Statewide Policy Framework (GSA 2009). Variables for scenario modelling were the proportional area of pine-forestry planting within the catchment (‘planting fraction’) and the width of hydrological buffer zones. Planting fractions used were 12%, 30%, 50% and 100%, while for each planting fraction four different buffer widths were modelled; 10 m, 20 m, 50 m and 100 metres.

This study does not seek to compare wetlands in catchments of plantation forest or native forest, nor does it seek to comment on the ecological change that may have occurred from conversion from native forest to pasture. The study works from the premise that wetlands in their current state support particular ecological character and seeks to provide insight into how future changes in hydrology may manifest in changes in this character.

For the purposes of this assessment, all forestry scenarios are compared with the surrogate baseline of a 100% pasture response. While the monitored pasture catchment included a small proportion (~10%) of deep rooted perennial cover incorporated in isolated paddock trees and linear shelter belts, no attempt is made in this study to reconcile the difference in runoff between pasture and a ‘mixed land use’. Runoff comparisons are made against a land cover comprising 100% pasture.

2. AIM AND OBJECTIVES

The central aims of this project were to explore the nature of the interaction between plantation forestry and the hydrology of small perched wetlands and in turn, identify any likely impacts on the ecological character of the wetland ecosystem. The investigation was intended to develop a preliminary understanding of the hydrology and ecology to inform the additional work required to underpin sustainable management of perched wetlands from a water resource management perspective. The questions examined for perched wetlands were informed by the potential management responses:

- What proportional level of forest development creates an impact on catchment hydrology that would potentially change wetland ecological character?
- Are buffers an effective means with which to manage hydrological impacts on wetlands resulting from plantation forestry?

Objectives required to meet these study aims were as follows:

- Develop a conceptual model for use in modelling perched wetland hydro-ecological function
- Install monitoring networks at representative sites for pasture, forest and native vegetation
- Calibrate surface water model parameter sets consistent with the observed behaviour under each land use
- Run modelled scenarios with different forest proportions and buffer sizes and investigate the changes to wetland hydrology
- Predict ecological impacts under the different planting fraction and buffer width combinations modelled.

3. REGION AND STUDY AREA DESCRIPTION

3.1. *FLEURIEU PENINSULA*

The Fleurieu Peninsula comprises the southern extension of the main spine of the block-faulted Mount Lofty Ranges. The Peninsula is devoid of any coastal plains on both its northern and southern coastlines except for small area between Victor Harbor and Goolwa. The central spine, or plateau, consists of two areas of uplands exceeding 300 m elevation, which are separated by lowlands forming the upper reaches of Inman Valley. The highest elevation of 440 m is located in the northern upland area in the vicinity of Spring Mount near Myponga. This area forms the headwaters of the Myponga, Hindmarsh and Inman River catchments, which are the three largest catchments of the Peninsula. The narrower southern upland area ends at Cape Jervis and is traversed by Range Road, which runs along the central ridge known as the Parawa Plateau at an elevation of 320–70 m. Surface water catchments are shown in Figure 1 along with their average slope.

The major land uses in the southern Fleurieu Peninsula region are grazing of cattle (beef and dairy) and sheep; plantation forestry (predominantly pine, with some farm forest ventures under bluegum); and, conservation (notably Deep Creek and Talisker Conservation Parks).

Pre-European vegetation in the region consisted largely of heathy woodland or heathy open forest (DEH 2007), with riparian and wetland species found within the drainage lines. Typical hill slope vegetation associations including Messmate Stringybark (*Eucalyptus obliqua*) over a sclerophyllous shrub understory. Both the Springs Road and Foggy Farm study sites contain examples of this vegetation type. Riparian vegetation was more likely to include typical wetland species of the region such as tea-tree and swamp gum.

Average annual rainfall is influenced by elevation and prevailing seasonal winds and ranges from about 500 mm per year along the narrow areas of the coast near Victor Harbor, Cape Jervis and Carrickalinga to over 850 mm per annum in the highest elevation areas centred on Mt Compass in the north and Parawa in the south (Fig. 2).

The geology and hydrogeology has been previously described in Barnett and Rix (2006), a reference which also introduces the different sub-types of Fleurieu Peninsula wetlands (see following section). The Fleurieu area is underlain by a variety of consolidated basement rocks and unconsolidated sediments that have exerted a strong control on the formation of the current landscape. Figure 3 shows that the central northern part of the Peninsula is underlain by the Barossa Complex. This has been exposed by erosion to form the central core of the Fleurieu, including the highest topography described above. The southern two thirds of the Peninsula are underlain by Kanmantoo Group meta-sediments with more recent alluvium in central locations (Barnett and Rix, 2006). The glacial infill sedimentary Permian Sands separates the two upland areas, with a mix of Adelaidean sedimentary rock found in smaller quantities.

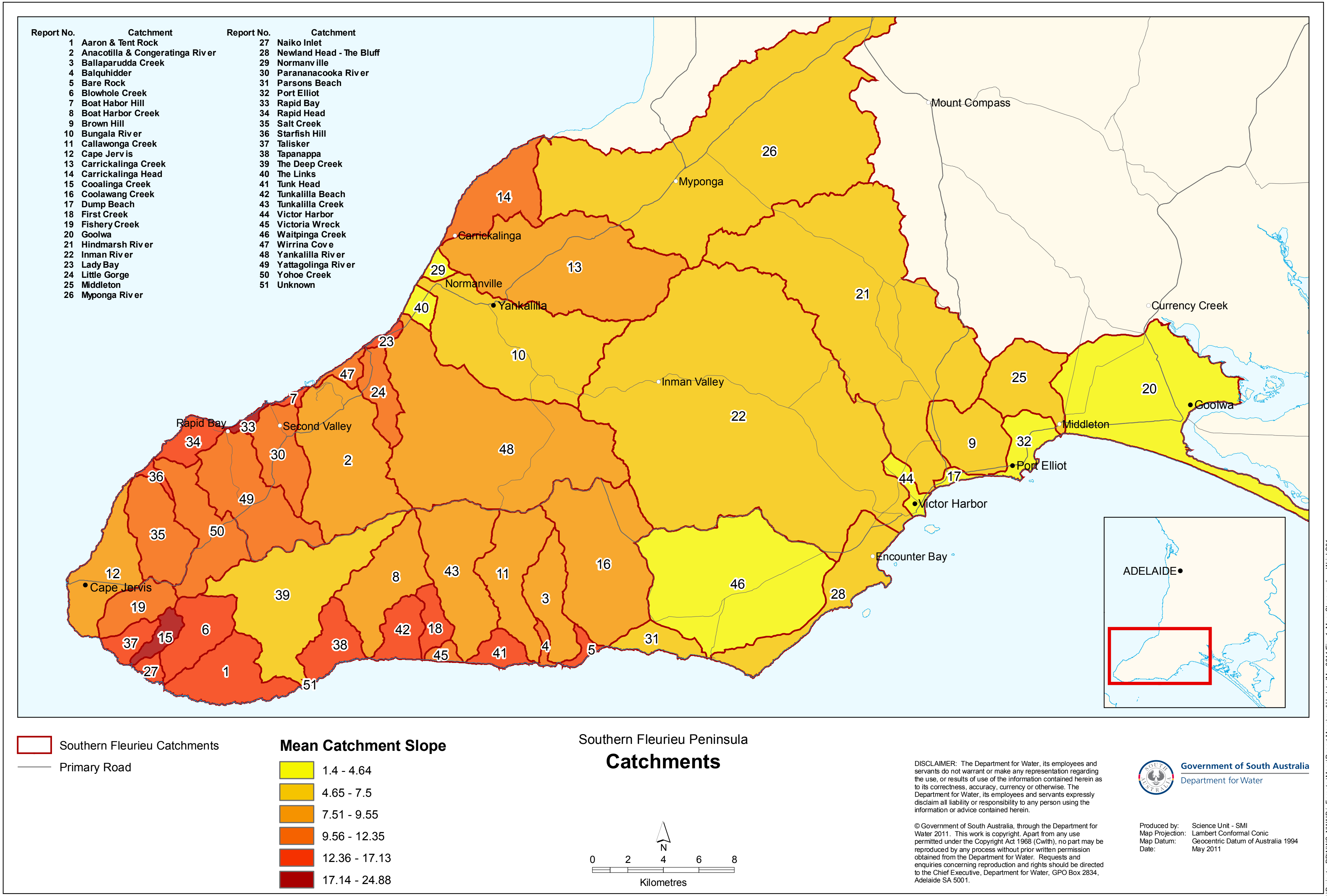


Figure 1. Southern Fleurieu Peninsula catchments with average slope

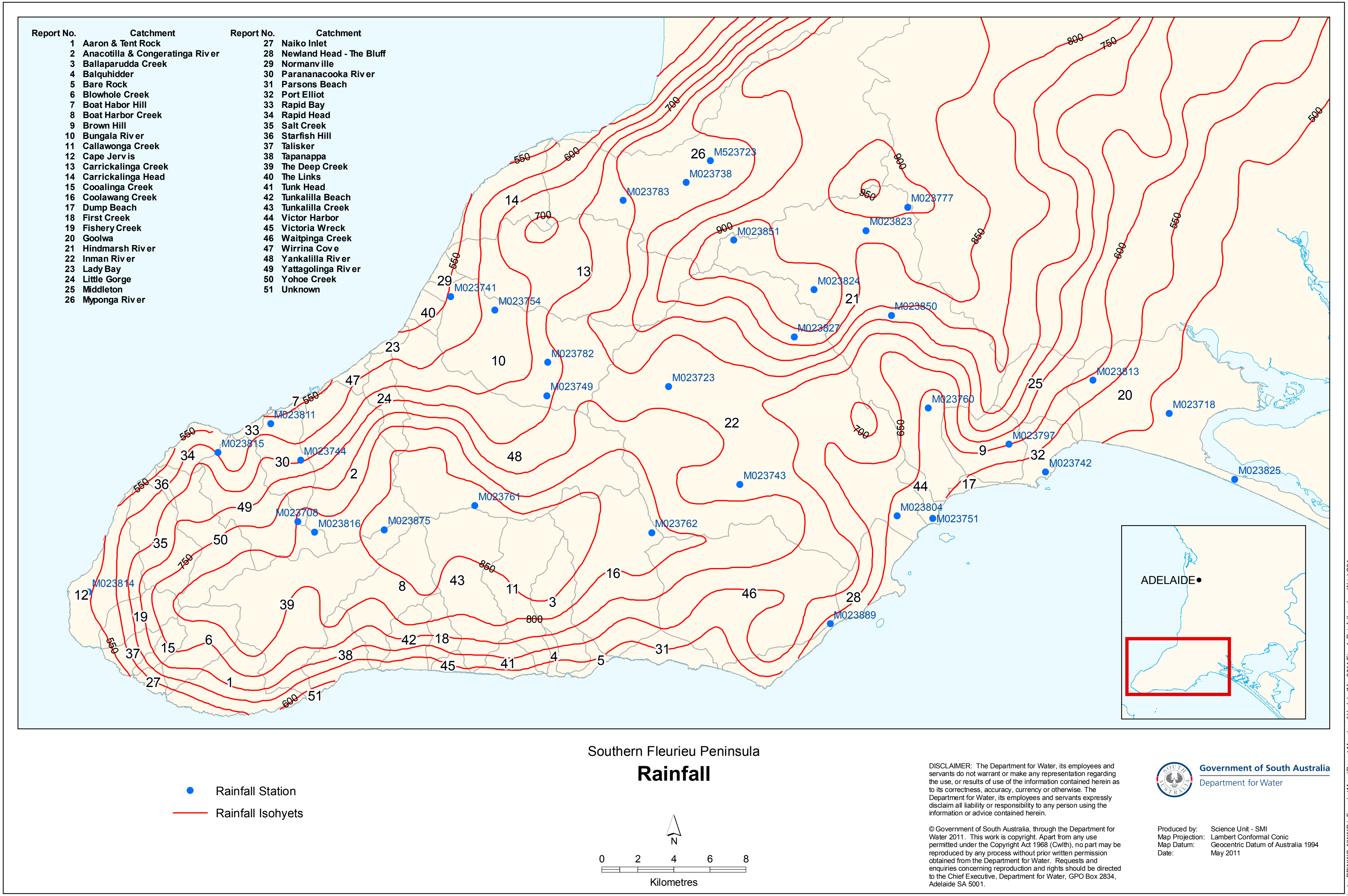


Figure 2. Rainfall isohyets and rainfall stations for the Southern Fleurieu Peninsula

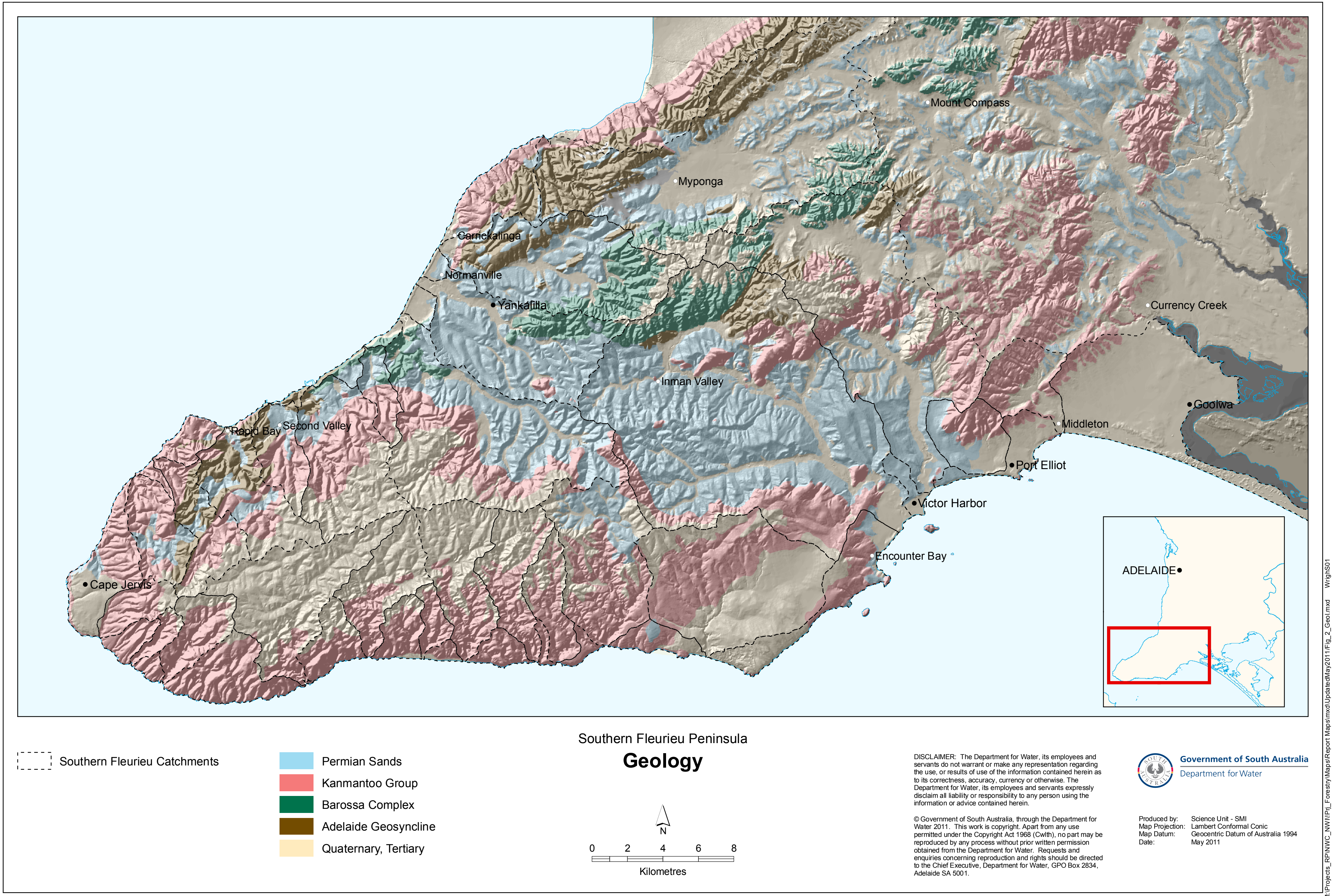


Figure 3. Geology of the Southern Fleurieu Region

3.2. WETLANDS OF THE FLEURIEU PENINSULA

3.2.1. WETLAND SUB-TYPES

The majority of wetlands found on the Fleurieu Peninsula are located within the two broad physiographic units described above: in drainage lines of the upper slopes directly adjoining remnant plateaux (Barossa Complex and Kanmantoo Group meta-sediments); and in valley floors of infilled glacial depressions (Permian Sands; Duffield and Milne, 2000, Barnett and Rix, 2006). The landscape setting in which wetlands are located influences the hydrogeology and ecological character of the wetland. Barnett and Rix (2006) used hydrogeology to define four separate wetland classifications, based on the distinct characteristics of groundwater across the landscape gradient within which wetlands are found. Perched wetlands and Permian Sands wetlands represent the endpoints on the continuum of conditions, with two intermediate classes (‘fractured rock’ and ‘transition’). A detailed description of the underlying hydrogeological processes supporting these different systems can be found in Barnett and Rix (2006) and Vanlaarhoven and van der Wielen (2009), while the following gives just a brief introduction to Permian Sands wetland processes before developing the function and character of perched systems in more detail. Only the hydrological processes for perched systems were considered in this investigation.

Wetlands underlain by Permian Sands usually occur in the lowest parts of the landscape, in valleys and depressions, where they are in direct contact with the regional watertable aquifer (Barnett and Rix 2006). As a result of these sandy soils, there is very little surface runoff and groundwater provides almost all of the wetland water requirements. Less than 20% of the wetlands of the Fleurieu Peninsula are considered to be Permian Sands wetlands (Barnett and Rix 2006). While the hydrological processes of the Permian Sands were not a focus of this investigation, data on watertable dynamics and vegetation distribution were incorporated in the study to increase the generality of vegetation modelling.

In contrast, perched wetlands are entirely dependent upon local hydrological processes and have no contribution from the deeper regional groundwater in the regional fractured rock aquifer. Perched wetland hydrology is dependent on rainfall–runoff or lateral sub-surface soil water movement processes from the soil profile above any clay layer (Fig. 23; after Barnett and Rix 2006). As a result of their reliance on local processes, perched wetlands are far more vulnerable to changes in hydrology in their immediate catchments due to land use change. These wetlands are also by far the most abundant across the landscape in the region (>75% - see Fig. 4). As a result of their numerical dominance, vulnerable hydrology and limited understanding of the supporting processes, perched systems are the focus of this investigation. The following section provides additional background on perched wetlands.

Plateau region (perched) wetlands

Wetlands are considered perched when they occur in drainage lines over clayey weathered basement, which can attain a thickness of up to 30 m (Barnett and Rix 2006). As a result, any losses from the drainage line by vertical infiltration are considered insignificant. No effective connection is believed to exist between the wetlands at the surface and the fractured rock aquifer beneath the weathered basement.

Perched wetlands are typically found on drainage lines high in the landscape adjoining remnant plateaux surfaces. These plateaux represent the land surface corresponding to a pre-Tertiary period peneplain that was uplifted during the Tertiary and Quaternary periods and subsequently deeply dissected (Campana and Wilson, 1953). Remnant plateau surfaces consist of soils overlying deeply weathered sediments, while slopes consist of relatively shallow soils overlying weathered rock. As remnant topographic highs, plateau areas typically form the catchment divide between major rivers. A result of this is that clusters of perched wetlands in the landscape, found within close physical proximity, occur within quite distinct drainage basins (Fig. 4).

Wetlands found within this physiographic setting are concentrated in three areas of the Peninsula (Fig. 4). Perched systems in the three areas have similar developmental and weathering histories and are associated with deeply weathered materials, but are underlain by different rock (compare Fig.3 and Fig. 4). A large majority of these wetlands are located in the southern area adjacent to the Parawa plateau where they overlie Kanmantoo Group meta-sediments (Barnett and Rix, 2006), consisting of siltstones and sandstones. As a result, this study has focussed on sites near to Parawa. Wetlands in this region are more common in streams draining to the south of the plateau, with relatively few wetlands located to the north of the Range Road watershed.

The other two small clusters of perched wetlands are both found further to the north of the Peninsula over Barossa Complex granites or gneisses (Barnett and Rix, 2006; see Fig.3). A cluster of perched swamps are found near to Clarke Hill, occurring across the catchment divide separating the Hindmarsh, Inman and Myponga Rivers and Carrickalinga Creek. The third major grouping is found slightly to the north-east on the slopes of Mount Cone, the majority being within the Myponga and Hindmarsh River headwaters.

A relatively small number of perched swamps are found in the upper Finniss River catchment north of Mount Magnificent and the upper Currency Creek catchment to the west of Mosquito Flat. These overlie Kanmantoo Group meta-sediments and Barossa Complex rocks respectively.

The character of perched wetlands reflects their landscape position in relatively broad valleys close to upper catchment divides. Many perched wetlands have formed on first or second order streams and have only small contributing catchments (often of the order of 100 ha). Topography as well as hydrology does not generally support extensive lateral development and wetlands tend to form with linear planform, closely aligned with watercourses. The nature of local geology, topography and climate creates conditions for a perched aquifer to form and it appears likely that some near, or shallow sub-surface flow barrier (such as an impermeable rock bar) prevents drainage in the downstream direction. This creates a storage that is poorly drained, prolonging retention time and allowing the wetland to develop.

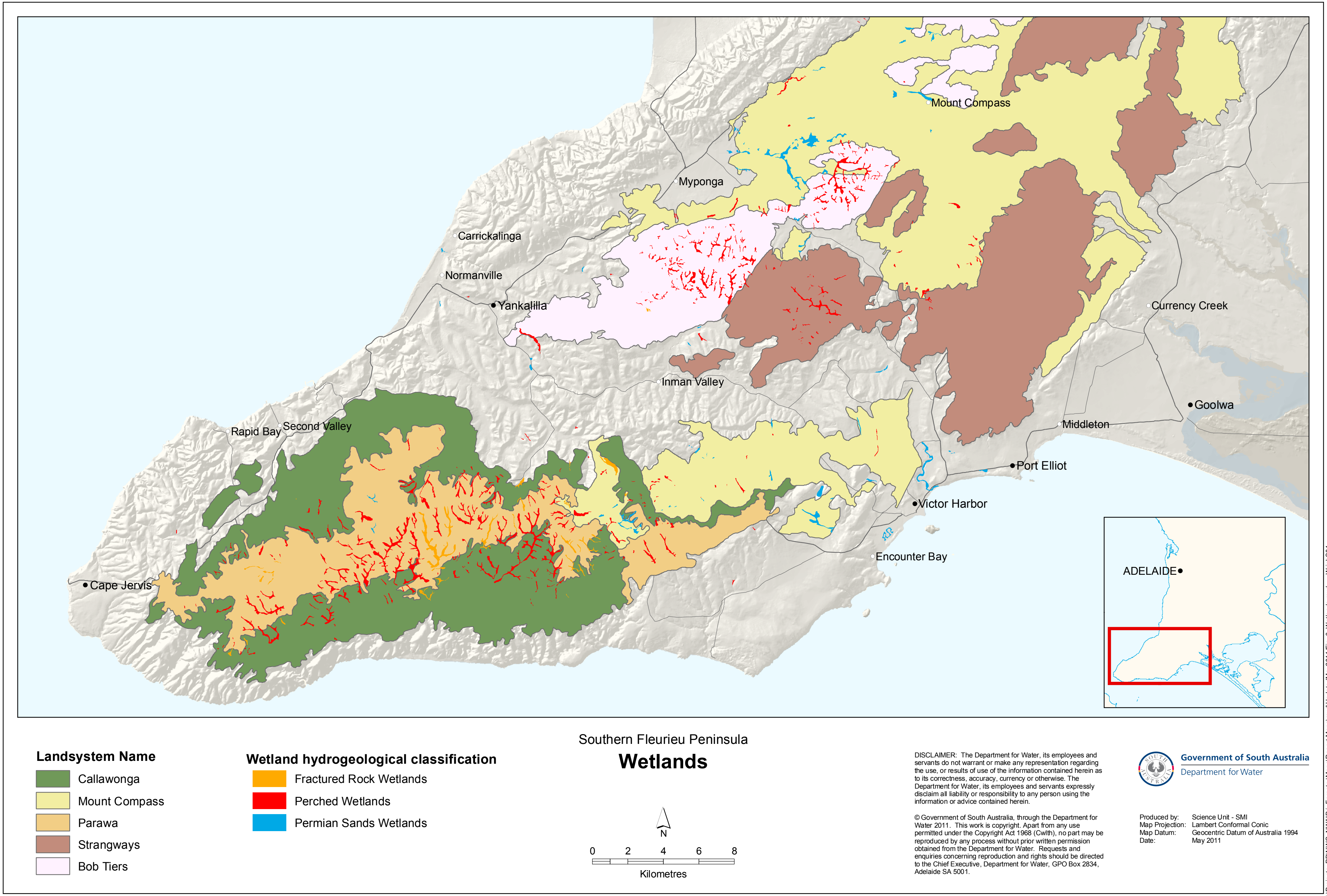


Figure 4. Wetland types and selected landsystems for the Southern Fleurieu Peninsula

3.2.2. ECOLOGICAL CHARACTER

Wetlands of the region are described as comprising dense vegetation that is 'reedy or heathy' in nature, growing on waterlogged soils typically associated with watercourses (Duffield and Milne, 2000). Owing to climatic conditions and physiographic setting, any inundation of wetland substrates is generally shallow and of short duration. Persistence of standing water habitat in particular is limited to perhaps a few centimetres depth of water over winter months, with the duration varying according to seasonal and longer climatic cycles. Peat formation is favoured under conditions of enduring saturation and surface water. The implication of the shallow nature of inundations for obligate aquatic species is to confine larger life forms to watercourses and deeper permanent pools in particular.

'Intact' or 'semi-intact' wetlands (structurally speaking) are described as comprising a mosaic of patches with varied structural form that merge into one another. Structural formations typically comprise up to three distinguishable layers (Duffield and Milne, 2000):

- The highest stratum is a medium–tall shrub layer, typically comprising one or more of the following species: *Leptospermum continentale* (prickly tea-tree), *L. lanigerum* (silky tea-tree); *Acacia retinodes* (swamp wattle); *Viminaria juncea* (native broom) *or Melaleuca decussata* (cross-leaved honey-myrtle). A eucalyptus overstory is typically not present, but isolated individuals or small stands of *Eucalyptus ovata* (swamp gum) or *E. cosmophylla* (cup gum) are recognised in Duffield and Milne (2000) as sometimes being found. Evidence presented in Harding (2005) suggests that *E. ovata* is a common species associated with wetlands, being present at more than half the sites surveyed as part of the Wetland Inventory for the region. This is a wetland fringe species and, as suggested in Duffield and Milne (2000), is usually observed in small stands or as isolated individuals.
- The intermediate stratum comprises a tall sedge and/or fern layer typically including the following species: *Lepidosperma longitudinale* (weeping tea-tree), *Baumea rubiginosa* (soft twig rush); *Gahnia sieberiana* (red-fruit saw-sedge); *Blechnum minus* (soft water fern); *Pteridium esculentum* (bracken fern); and, *Baumea tetragona* (square twig rush), which were all present at more than 70% of wetlands surveyed in Harding (2005). In many cases the sedge layer is the highest stratum, the shrub layer being absent.
- The lowest stratum groundcover comprises a variety of herbaceous species, grasses or low-lying sedges including: *Isolepis inundata* (swamp club rush); *Eleocharis gracilis* (slender spike-sedge); and *Patersonia occidentalis* (native iris) (Duffield and Milne, 2000; Harding 2005). Many groundcover species are annuals and in some cases exotics, including the most commonly found introduced plant found of wetlands according to the Wetland Inventory (Harding 2005), *Holcus lanatus* (Yorkshire fog grass). This species was especially prevalent at the wetland in the pasture field site for this study.

3.2.2.1. Plant functional groups in perched wetlands

Casanova and Brock (2000) developed a functional classification for wetland plants based on their different water regime requirements, preferences, growth habit and growth response to inundation. Common wetland plants of the Mount Lofty Ranges were assigned to these categories for use in water allocation planning in the region (Casanova *et al.* unpub.; Vanlaarhoven and van der Wielen 2009).

A description of the water requirements of the different plant functional groups is presented in Vanlaarhoven and van der Wielen (2009), which separates the groups between semi-aquatic and true aquatics as follows:

- Submerged emergent (SE**) – e.g. *Typha domingensis*; *Triglochin procerum*
- Amphibious fluctuation tolerator emergent (Afte*) – mostly sedges such as *Cyperus* or *Juncus*
- Amphibious fluctuation tolerator woody-habit (Aftw*) - typically *Leptospermum* or *Melaleuca*
- Amphibious fluctuation responder plastic growth form (Afrp*) –e.g. *Villarsia umbricola*
- Amphibious fluctuation tolerator low growth form (Aftl*) – e.g. *Isolepis* spp
- Terrestrial damp (Tdamp*) – e.g. *Eucalyptus ovata*
- Terrestrial dry (Tdry) – all terrestrial plant species.

Where:** = true aquatic; * = semi-aquatic; and no asterisk indicates no specific environmental water requirement. Classes from Vanlaarhoven and van der Wielen (2009).

Analysis of vegetation communities present in a given wetland using these classes can provide an indication of the water regime to which the plant community has adapted. Hence, by gaining an understanding of the nature of perched wetland vegetation communities this also provides insight into their hydrology, a technique known as calibration (ter Braak 1995).

Presence/absence vegetation-species data, collected across 156 perched wetlands in ground-truthing surveys for the SAWID project (Harding 2005), were transformed to plant functional-group counts. The proportional representation (number of species in the total sample species richness) in each functional group was determined, giving the proportion of total species richness falling within each functional group for each wetland.

A boxplot of the proportional plant functional group representation data indicates amphibious fluctuation tolerators of emergent habit (Afte) had the highest representation, with a median proportion of 0.6 of perched wetland species richness across all surveys (Fig. 5). Other high proportion functional groups were Aftw (amphibious fluctuation tolerators of woody growth habit. In Fleurieu Swamps these are typically the tea-tree species mentioned above) and the two terrestrial functional groups Tdry and Tdamp. Wetlands recording zero species from these four groups are rare (seen as outliers in Fig. 5).

Afte plant functional group comprises largely sedge species. Vanlaarhoven and van der Wielen (2009) classify Afte and Aftw (referred to therein as ‘ATe’ and ‘ATw’ respectively) as semi-aquatic functional groups with establishment and growth requiring damp soil to shallow water (see Vanlaarhoven and van der Wielen 2009; Appendix E Part 1, p. 71). Species classified as Tdamp require saturated soil to establish, while Tdry are terrestrial species with no specified environmental water requirement.

Functional groups classified by Vanlaarhoven and van der Wielen (2009) as true aquatics (Amphibious fluctuation responders of plastic growth habit [Afrp], submerged emergent [SE], submerged k-selected [Sk] and submerged r-selected [Sr]) occur in generally low proportions in the sample dataset. Species within these functional group categories require saturated soils to shallow standing water (Vanlaarhoven and van der Wielen 2009). Submerged emergent [SE] species were not present at a little over 25% of perched wetlands surveyed and Sk was rarely observed (Fig. 5). Sr species were recorded in such low numbers in surveys that they are not displayed in the figure.

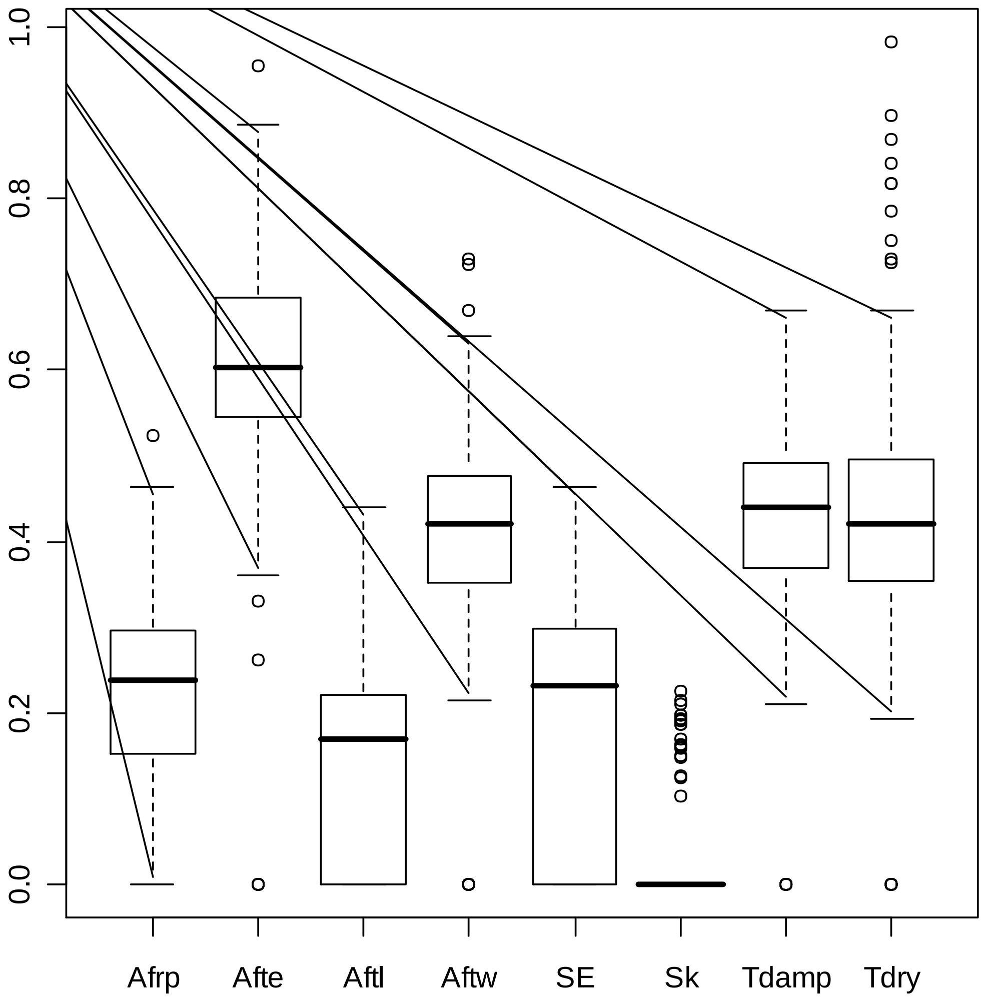


Figure 5. Proportional abundance of different vegetation functional groups across a sub-sample of 156 perched wetlands located on the southern Fleurieu Peninsula (data from Harding, 2005).

The relative proportions of the different functional groups suggests most perched wetlands have habitat that is suited to semi-aquatic functional groups—damp to saturated for much of the year. Not all perched wetlands provide habitat with extended periods of surface soil saturation or standing water suitable for true aquatic species based on this survey data and where present they tend to form a relatively low proportion of species richness.

3.2.3. BIODIVERSITY VALUES

Wetlands are significant biodiversity assets in the region and many of their floral and faunal constituents are of high ecological value (Duffield and Milne, 2000; Harding 2005). Calls for conservation of Fleurieu wetland ecosystems can be traced to at least 1982 (Davies, 1982) and shortly thereafter, a report into the conservation of native vegetation in the State found that almost half of a large sample of plants of conservation significance in the southern Fleurieu region were confined to upland wetlands (Lang and

Kraehenbuehl, 1987). The identified biodiversity values of a subset of wetlands in the region, which came to be known as Swamps of the Fleurieu Peninsula, led to the nomination (Duffield and Milne, 2000) of these wetlands as a matter of national environmental significance under the *Environment Protection and Biodiversity Conservation Act (1999)*. More recently, Harding (2005) recorded 742 vegetation species during wetland field surveys, of which 80% of species were indigenous and 30% of conservation significance, demonstrating the habitat values of these systems.

Although much prior work has concentrated on mapping and describing vegetation within wetlands, work on other biota has also been undertaken. Aquatic macroinvertebrates are not particularly well known, but are considered to be diverse in Mount Lofty Ranges streams (Armstrong *et al.*, 2003). It would seem reasonable to assume that watercourses within wetland habitat are likely to have invertebrate biodiversity at least equal to watercourses with cleared catchments. Native fish have also been recorded from Fleurieu wetlands (Harding, 2005).

Wetlands in the region are also noted for the numbers of endangered and listed species of vertebrates found permanently, seasonally or occasionally within their extent. The most well known, although not obligate, inhabitant of wetlands in the region is the EPBC-listed (critically endangered) Mount Lofty Ranges Southern Emu-wren. Details of other bird and mammal species known from Fleurieu wetlands can be found in Harding (2005).

Littlely (1998), in a survey of twelve swamps of the region, suggests that vertebrate diversity was highest in those which were intact—whilst structurally degraded wetlands lack mammal species, although they can still provide important bird habitat. The same authors found plant diversity was lowest in largely untouched swamps, with a 'lightly grazed' swamp having the highest. This finding is consistent with a number of prior researchers who have suggested that lack of disturbance, in particular fire, reduces the overall diversity of wetlands in the region as competitively superior species become dominant (Littlely, 1998, Duffield and Milne, 2000).

The recognised biodiversity value of wetlands of the region perhaps reflects the highly modified landscapes of the region. Along with remnant patches of native vegetation on hilltops, the difficulties associated with developing waterlogged habitats have likely contributed to their current value as remnant habitat as development for production has not been viable.

Evidence presented in Harding (2005) and Littlely (1998) suggests that wetlands with peaty soils have higher species richness than analogous systems with mineral-dominant soils, although no functional relationship has been suggested for this observation. As submerged functional groups appear to be found in relatively few perched wetlands, those which are wettest could be expected to support additional species richness. Hence wetland hydrology may control not only the presence of peat (which requires permanent saturation to form), but also higher plant species richness through provision of additional habitat diversity.

3.2.4. WETLANDS AND CATCHMENTS OF PRE AND POST-EUROPEAN LANDSCAPES

Considerable information on changes to vegetation, catchment land use, sedimentation rates and hydrology has been gleaned from analysis of sediment cores taken from wetlands on the southern Fleurieu. The following discussion relies on two recent publications in which the influence of climatic fluctuations and both aboriginal and European catchment land management practices on wetland geomorphology and vegetation communities are integrated (Bickford and Gell, 2005, Bickford *et al.*, 2008).

Perched wetlands have been present as an element of the Fleurieu Peninsula landscape for over 8000 years (Bickford and Gell, 2005). During the period 7000–5000 years before the present time, climatic conditions were wetter than is currently the case, as shown in sediment cores by high rates of peat accumulation (Bickford and Gell, 2005). During this period, conditions within a perched wetland near to the current study sites were permanently moist and lateral expansion of tea-tree (*Leptospermum* sp.) is thought to have occurred (Bickford and Gell, 2005). The presence of *Typha* pollen in the record from this period indicates the climate was wetter than the late Holocene through to the present day, though the dominance of the pollen of two contemporary sedges (*Carex* and *Baumea)* suggests that the depth of water was not significantly greater than those of today (Bickford and Gell, 2005). *Typha* can tolerate water to depths of around 1.5 m, but persistent water of any depth would have precluded the other sedge species from the wetland.

Later in the Holocene period the climate shifted to a drier phase, leading to low rates of net sediment accumulation, possibly including a loss of accumulated peat through deflation[1] (Bickford and Gell, 2005). Catchment vegetation during drier periods was characterised by Eucalypt woodlands similar to those found in extant remnant patches of native vegetation in the region, while no current analogue exists for the *Allocasuarina* association present in the relatively wet early Holocene (Bickford and Gell 2005).

Land management practice by aboriginal people is thought to have involved frequent low intensity burning of wetlands and associated catchments, which did not result in any detectable changes in species associations of wetlands or their catchments (Bickford and Gell, 2005). In contrast, changes to catchment vegetation and therefore hydrology, has been profound since European settlement of the region in 1839 with phases summarised in Bickford *et al.* (2008):

- Early European land uses were native forestry, cropping and grazing. During the period high rates of soil loss resulted from the practice of ploughing hillslopes oblique to contour lines (Twidale 1971) and high intensity fires compared with those occurring under aboriginal management regimes. Rates of sedimentation during this period were 0.75 cm/y, higher than at any time in the available record.
- During the late 19th and first half of the 20th centuries land use intensified, including increased catchment firing for fodder improvement on 2–5 year cycles. This resulted in the loss of fire-sensitive understorey species from catchments.
- Widespread clearance occurred in highland forests and scrub around 1950 with increased knowledge of nutrient deficiencies in soil (including trace element deficiencies), development of suitable fertilisers, improved pasture species and establishment techniques, superior techniques for clearing dense eucalypt scrub and larger more powerful machinery. This resulted in the widespread displacement of native deep rooted perennial vegetation to graminoid species.
- Decreased sedimentation rates have been observed (0.2–0.3 cm/y) Post 1950, concurrent with increasing quantities of *Pinus* pollen through expanding forestry of the genus. Declines in sediment accumulation could be due to decreases in sediment input, or through decreases in the build-up of vegetative matter, possibly an indication of lower productivity (Bickford and Gell, 2005).

In general, major changes to the vegetation of wetlands described by Bickford and Gell (2005) and Bickford *et al.* (2008) coincided with greatly increased periods of erosion. Concurrent with these changes in the composition of the wetland sediments were changes in vegetation. From fringing *Acacia* and wetland species such as Cyperaceae*, Adiantum aethiopicum* and *Gleichenia microphylla* (maidenhair and coral fern respectively), dominant vegetation shifted to *Blechnum* spp, newly colonising *Leptospermum* spp (tea-tree) and expansion of the Cyperaceae. Both of these observations may have been driven by profound changes to the hydrology resulting from European catchment management

[1] The erosion by wind of the peat surface

practices of the day. Contributing activities described included both clearance and increases in burn frequency and intensity represented by large quantities of macro-charcoal in the record[2]. Changes to the wetlands may still be evident today. Bickford *et al.* (2008; p. 434) noted a shift in the character of the wetland from a *'small silty wetland, more creek-like than it is today'* to a more peat-rich soil *'characteristic of those formed in sedge and fern swamps'* either side of the depositional phase associated with clearance. The presence of peat would be consistent with increased levels of wetness and reduced erosion over the post clearance period.

Anecdotal information also suggests that in the southern Fleurieu region land was frequently burnt by settlers to regenerate sheep feed by lighting fires on the catchment divide allowing them to run to the coast with the aid of north wind. Cutting firewood and posts was believed to be a frequent economic activity, along with burning to cause a regeneration of wattle for sale. These activities would have altered the composition and age structure of native vegetation and depending upon scale, would have possibly influenced downstream hydrology (GSA, 2005).

Based on the official records, photographic evidence of large-scale native vegetation clearance during 1950s and 1960s and oral history, vegetation on the southern Fleurieu Peninsula has been in constant change over the last 50–100 years. In the Deep Creek catchment area, the first available photograph dating from 1949 indicated that 85% of the catchment was covered with native vegetation and that the remaining 15% has been cleared, presumably for grazing. Over the period from 1949 until the first forestry planting in 1972, these proportions changed; with native vegetation cover reduced to 40% and cleared land increasing to 60%. Over the period from 1972 until 1994, 67% of total land was converted to plantation forestry, primarily *Pinus radiata* (GSA, 2005). Some of the concurrent changes to wetland and catchment character are recorded in the pollen within sediment cores and discussed in Bickford and Gell (2005) and Bickford *et al.* (2008).

Evidence suggests that wetlands have been a part of the landscape of the Fleurieu Peninsula since shortly after the end of the last ice age. Wetland sedimentation rates and vegetation both within and surrounding the wetlands, have changed in response to catchment land management and climate since initiation. Some lateral expansion may have occurred during wetter climatic periods, but water depths are unlikely to have ever been limiting to the presence of species typical of wetlands today. The deposition of large amounts of sediment concurrent with more intensive land uses after European settlement may have changed wetland character and peat accumulation rates. Bickford and Gell (2005) report identifying pollen to depths of 1.8 m in sediment cores. The current study found little evidence of deposition of sediment above soil forming in situ within wetland sediments greater than around 60 cm at any of the study wetlands. This suggests a large degree of variation is present in the depositional characteristics of perched wetland soils likely attributable to local factors and land management history.

3.3. REGIONAL HYDROLOGY

A regional hydrological data review and analysis was performed at the initial stages of the project and is presented herein to provide additional context. Summary data are presented in this chapter with additional detail presented in Appendix E. This data analysis was done before the commencement of the data collection from the stations established for this project, hence the analysis is based on data from monitoring stations established before March 2008 (Fig. 8).

2 Very little charcoal was observed in soil descriptions at the three focus catchments in the current study and given the Bickford et al. studies cited were based on one or two soil cores, it is possible this may have been localised to the study site concerned.

3.3.1. RAINFALL AND EVAPORATION DATA

Rainfall is highly variable from year to year and between summer and winter periods, although the southern Fleurieu is recognised as one of the few areas in the state with relatively consistent rainfall, especially over the winter months when evaporation is also low. Mists and fogs are a common occurrence in the Parawa Plateau area and the typical extent of these has been anecdotally observed to coincide with areas where stringybark woodland is found (P. Filsell (Landholder, The Wither Swamp) 2007, pers. comm.). There is also significant variation depending on the location within catchments, with a noticeable decreasing rainfall gradient generally observed from catchment headwaters towards the coast.

The distribution of rainfall stations is concentrated along the spine and town centres of the Peninsula (Fig. 2). The distribution is particularly sparse along the southern coastal areas. Since rainfall is greatly influenced by topography, the lowest and highest rainfall tends to be located at the corresponding elevation within the catchment. While there is significant variation in rainfall over the region, regional similarities exist with winter rain predominating centred on June and July, with November to March being relatively dry (Fig. 6).

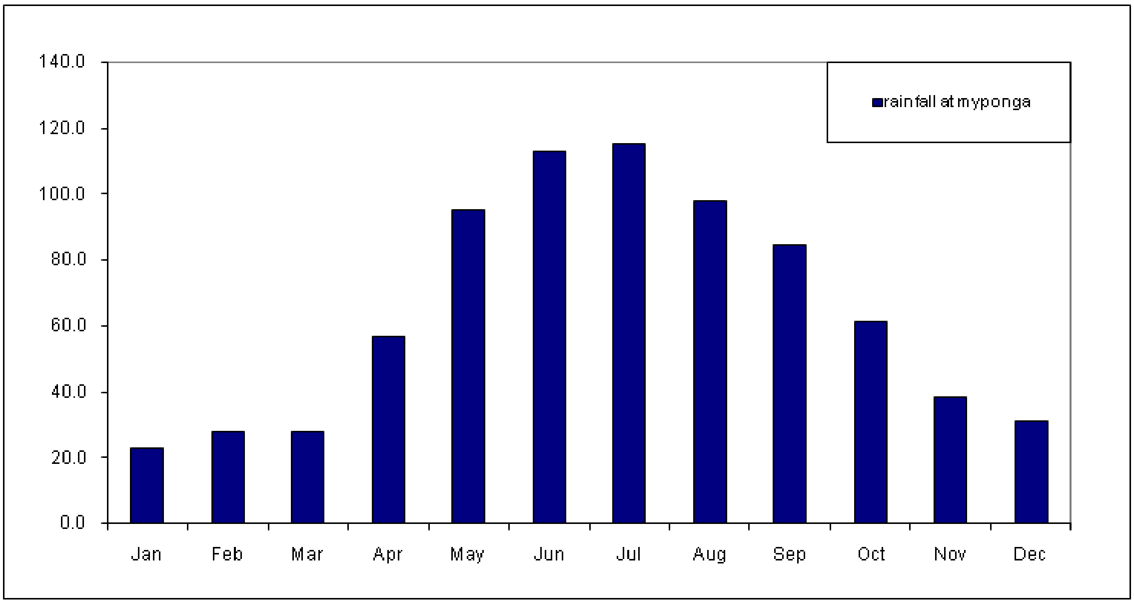


Figure 6. Monthly rainfall variability (in millimetres) at Myponga (station no: M023783)

The availability of daily evaporation data is limited, particularly in comparison to rainfall data. The nearest available Class A pan evaporation data for the study region is Myponga (station number M023783). Figure 7 indicates the mean annual evaporation for the period 1989–2006 was over 1300 mm and suggests a negative trend over the period. While mean monthly values for evaporation are generally used in the case of a lack of daily data. The limitation with this approach is that the large day-to-day variance in evaporation that often occurs during spring and autumn are not represented. This can result in large errors in daily time-step hydrological modelling. Estimates of evaporation across the topographic gradient for the region are a major hydrological data gap. Of particular interest would be variations in evapotranspiration rates to the north and south of Range Road in the vicinity of Parawa. Catchments with a southerly aspect support numerous swamps, while the northern, more exposed catchments feature relatively few swamps (Fig. 4). A rain-shadow effect to the north of the Parawa Plateau is suggested by the isohyets in Figure 2 and the combination of this and higher evapotranspiration in catchments with more northerly aspect are likely the major factors influencing the development of wetlands in the study region.

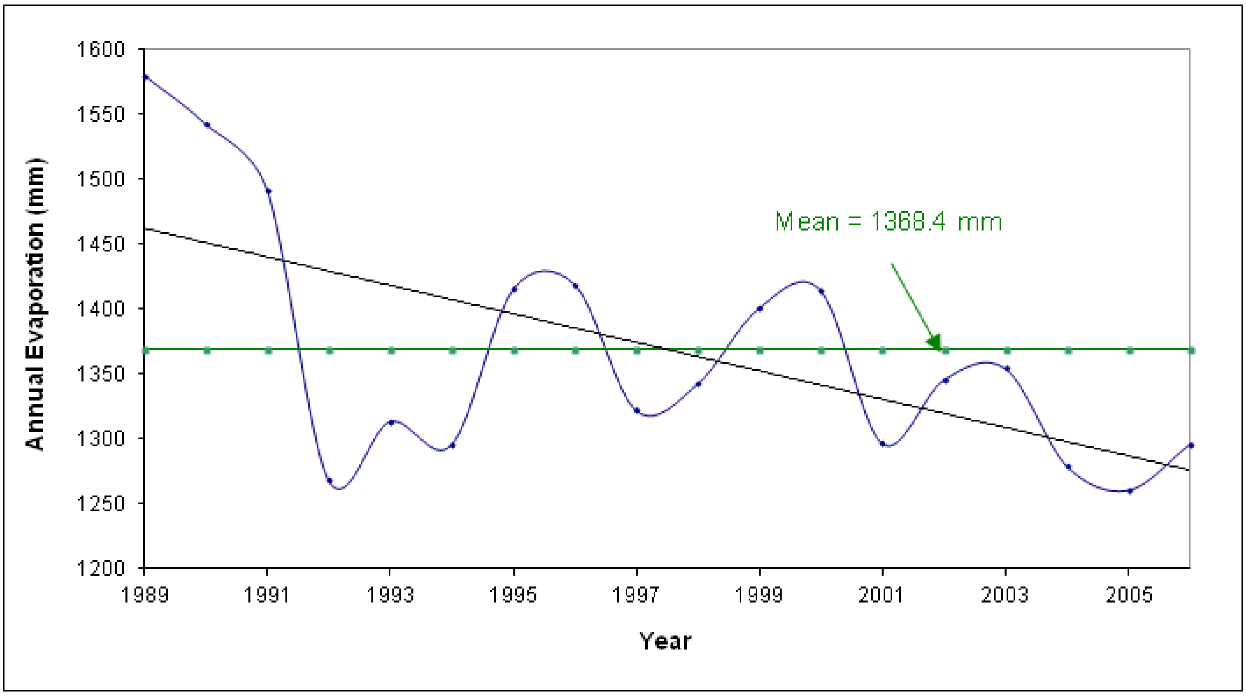


Figure 7. **Annual Class A pan evaporation in the region measured at Myponga (blue line), mean value for the period (green line) and linear trend (black line)**

3.3.2. STREAM FLOW DATA

Gauging stations on the Fleurieu Peninsula are very limited in number and stations with a long term period of record are concentrated in the central Fleurieu glacial infill valleys. Streamflow data is of limited duration for the study region, presenting limited opportunity for data analysis. The longest records relevant to the study area are for Boat Harbor Creek, Deep Creek and Ballaparudda Creek catchments (Fig. 9), which have catchment areas of 12.7, 10.1 and 8.6 km^2 respectively. Appendix E presents flow duration curves for each of these catchments. Both Upper Deep Creek and Boat Harbor Creek are intermittent at the site of the gauge, ceasing to flow for short periods over late summer – autumn, while Ballaparudda Creek had been perennial over the period of monitoring (2006–08).

Figure 8 shows the location of all existing gauging stations in the region before this project commenced, while gauging stations established during the current project are shown in Figure 9. All catchments with available data used for hydrological modelling in this study are presented in Figure 10.

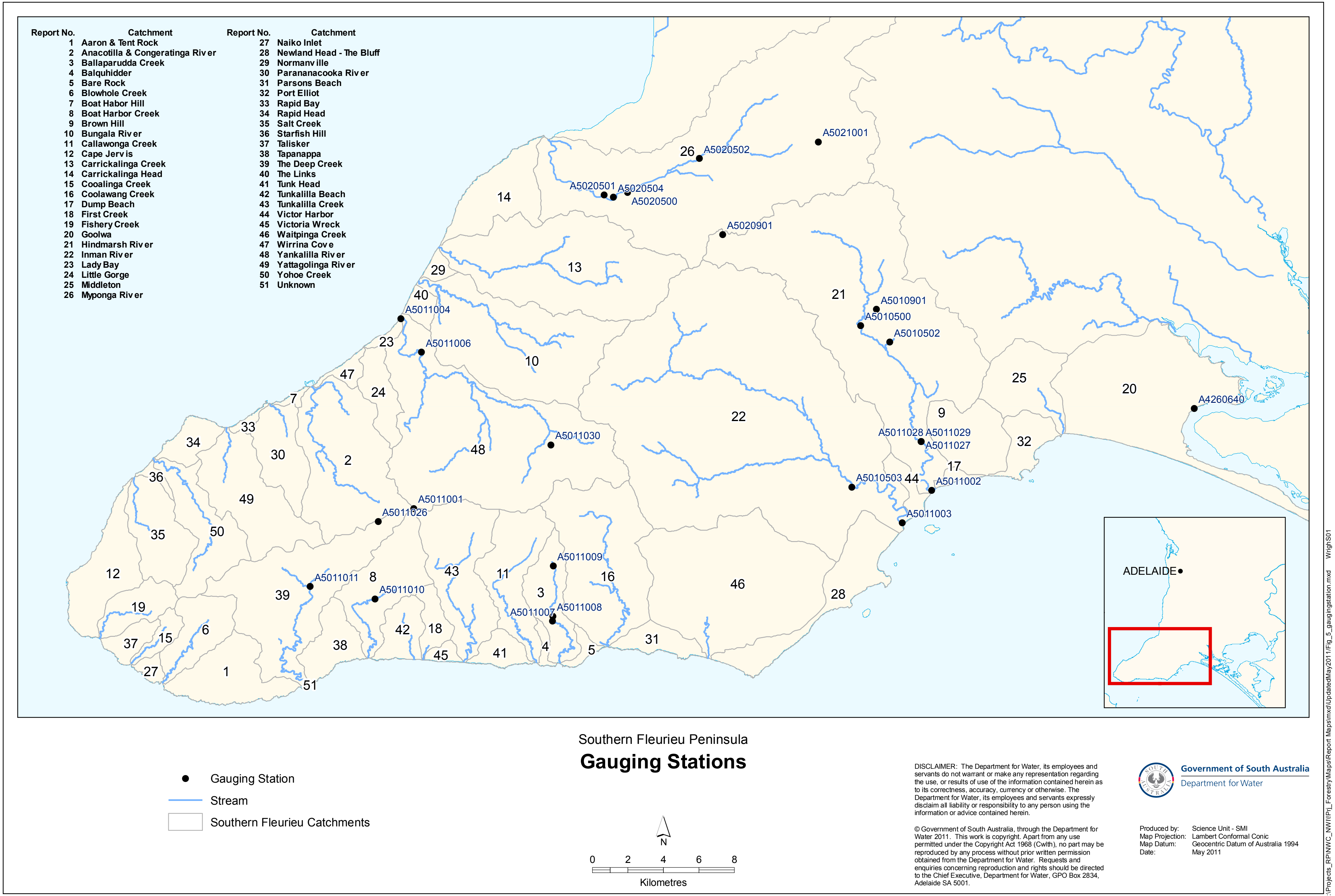


Figure 8. Southern Fleurieu Peninsula streamflow gauging stations in 2008

4. METHODOLOGY

4.1. *FIELD PROGRAM*

4.1.1. FIELD SITE CHARACTERISTICS

The field component of the project involved three small catchments located in the southern Fleurieu Peninsula region (Figs 9 & 10):

1. Foggy Farm (92 ha)
2. Wither Swamp (151 ha, although only the upstream gauge at Wither Swamp was used for modelling the pasture response, an area of 18.6 ha (Sect. 4.3))
3. Springs Road Native Forest Reserve (88 ha) (Figs 9 & 10; Appendix H).

The three catchments were under contrasting land uses and each supported a perched wetland. As shown in Appendix H, Foggy Farm is under pine plantation with a narrow buffer (typically 5 m) of pasture between the forest and wetland; Wither Swamp upstream catchment is under pasture; Springs Road Native Forest Reserve catchment is remnant stringybark (*Eucalyptus obliqua*) forest, but comprised around 12% pasture upstream of the wetland with a number of small onstream farm dams.

Selected land use history and physiographic characteristics of field sites is listed below. Sites were of similar elevation, but had a difference of over 30% in mean slope between the steepest and shallowest sites, although the range of values is typical for the region (compare Fig. 1).

Table 1. Selected field site characteristics

Site name and area	Mean elevation *(m)*	Mean slope *(°)*	Land use history	Current land use
Foggy Farm 92 ha	312	5.9	Cleared 1940s, grazing and cropping till pine planted from 1989	Pine plantation
Springs Road 88 ha	339	6.8	Largely undisturbed remnant native vegetation – Stringybark woodland Upstream area cleared, possibly in the 1940s	Conservation 76 ha Grazing 12 ha
Wither Swamp 150 ha	315	4.6	Cleared in 1940s for agriculture	Grazing (sheep and cattle)

4.1.2. INSTRUMENTATION AND FIELD SURVEY

Sites investigations employed a multi-disciplinary approach focussing on hydrology, soils and vegetation characteristics. The location of all deployed monitoring instruments is shown in Appendix H.

4.1.2.1. Continuous and opportunistic monitoring

Hydrological monitoring stations established under this project were deployed in late spring 2008, enabling rainfall and stream flow data for one full year to be used in analysis (2009 – see Sect. 5.1). Stations established for the purposes of this study are shown in Figure 9, while Appendix H presents an aerial photograph of each site indicating the placement of instrumentation used in this study. Data from each of these stations were used for hydrological modelling and are presented in Section 4.3 and 5.1. Each of the study catchments was instrumented to measure the following parameters:

Rainfall – at each study site a tipping bucket rain gauge (pluviometer) was installed prior to October 2008 (Appendix H). Rainfall intensity data at five minute intervals were available for all three sites.

Streamflow out of the wetland – continuous streamflow and salinity data was recorded from two culverts (Foggy Farm; Wither Swamp) and a purpose built weir (Springs Road).

Volumetric soil moisture content – instruments employing frequency domain reflectometry (FDR) to measure the volumetric water content of soil profiles were installed to obtain continuous (Enviroscan™) and point readings (Diviner 2000™) at multiple points around wetlands and catchments (Appendix H). Instruments were manufactured by Sentek (Stepney, South Australia).

Groundwater level in wetlands – hourly logged water depth sensors were installed in hand augered peizometers in all wetlands. Instruments were non-vented Levelogger Gold™ 10 m (manufactured by Solinst). Barometric pressure corrections were made using a Barologger™ deployed at each site. Loggers were downloaded quarterly and manual check groundwater levels were taken and verified against logged values.

In addition to streamflow leaving the wetland, Foggy Farm and Wither Swamp were fitted with stations to gauge inflows to the wetland from upstream, but no suitable site was identified at Springs Road for this purpose. Wither Swamp and Foggy Farm were monitored for farm dam levels and pumped dam offtake volumes. One A Class pan evaporimeter was installed at a location central to the three sites, but instrument failure during early deployment meant that data were not able to be used for modelling.

4.1.2.2. Field survey data

Additional field survey work to characterise behaviour were undertaken as follows:

Detailed soil survey and mapping – detailed soil survey work was conducted across the Foggy Farm and Wither Swamp catchments, which consisted of descriptions of hand-augured soil profiles and corresponding sites. Stereoscopic air-photo interpretation and soil profile descriptions were then used to develop soil landscape map unit boundaries, corresponding descriptions and land and soil attribute classifications for the Foggy Farm catchment (map unit descriptions, main soil groupings, geological and topographical descriptions and a description of main features, are given in Appendix D). Owing to project time constraints, map unit boundaries were not developed for the Wither Swamp catchment, nor was detailed soil survey work conducted at the Springs Road catchment.

Soil morphological and chemical characterisation – soil pits along a toposequence, from plateau or summit surface to wetland, were excavated at each catchment. Each excavated soil profile and corresponding sites were described in detail according to national standards (National Committee on Soil and Terrain 2009). Samples from each described soil horizon were then taken and subjected to comprehensive chemical analyses, also following standard chemical procedures (Rayment and Lyons 2010). Four sites were characterised at the Wither Swamp catchment (remnant plateau; mid slope; lower slope; wetland); four at the Springs Road catchment (remnant plateau; upper slope; lower slope; wetland) and five at the Foggy Farm catchment (remnant plateau; upper slope; lower slope; wetland margin; inner wetland). Full analyses, interpretations and reporting of these data have not yet been performed owing to project time constraints. Descriptions of each soil characterisation site are given in Appendix D.

Soil hydraulic characterisation – selected soil horizons and substrate layers from each excavated soil characterisation site (see above) were tested in situ for infiltration rate using large rings and locally sourced water, from which soil hydraulic conductivity rates were calculated. Corresponding samples were taken for measurement of volumetric moisture content. Soil strength tests were also conducted using a penetrometer (Appendix F).

Vegetation cover and abundance – multiple cover and abundance transects were conducted at each site. (Appendix I)

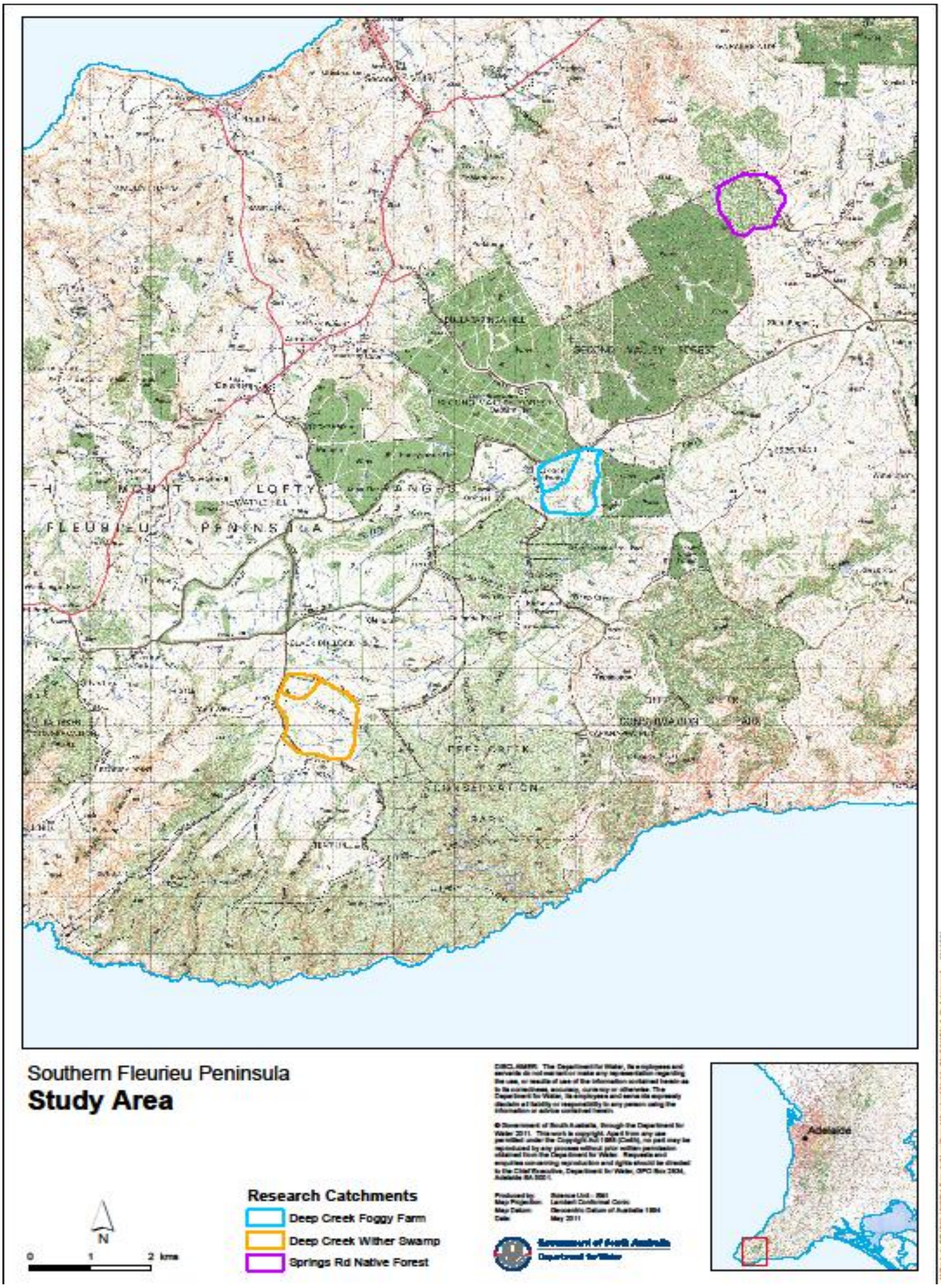


Figure 9. Location of study area

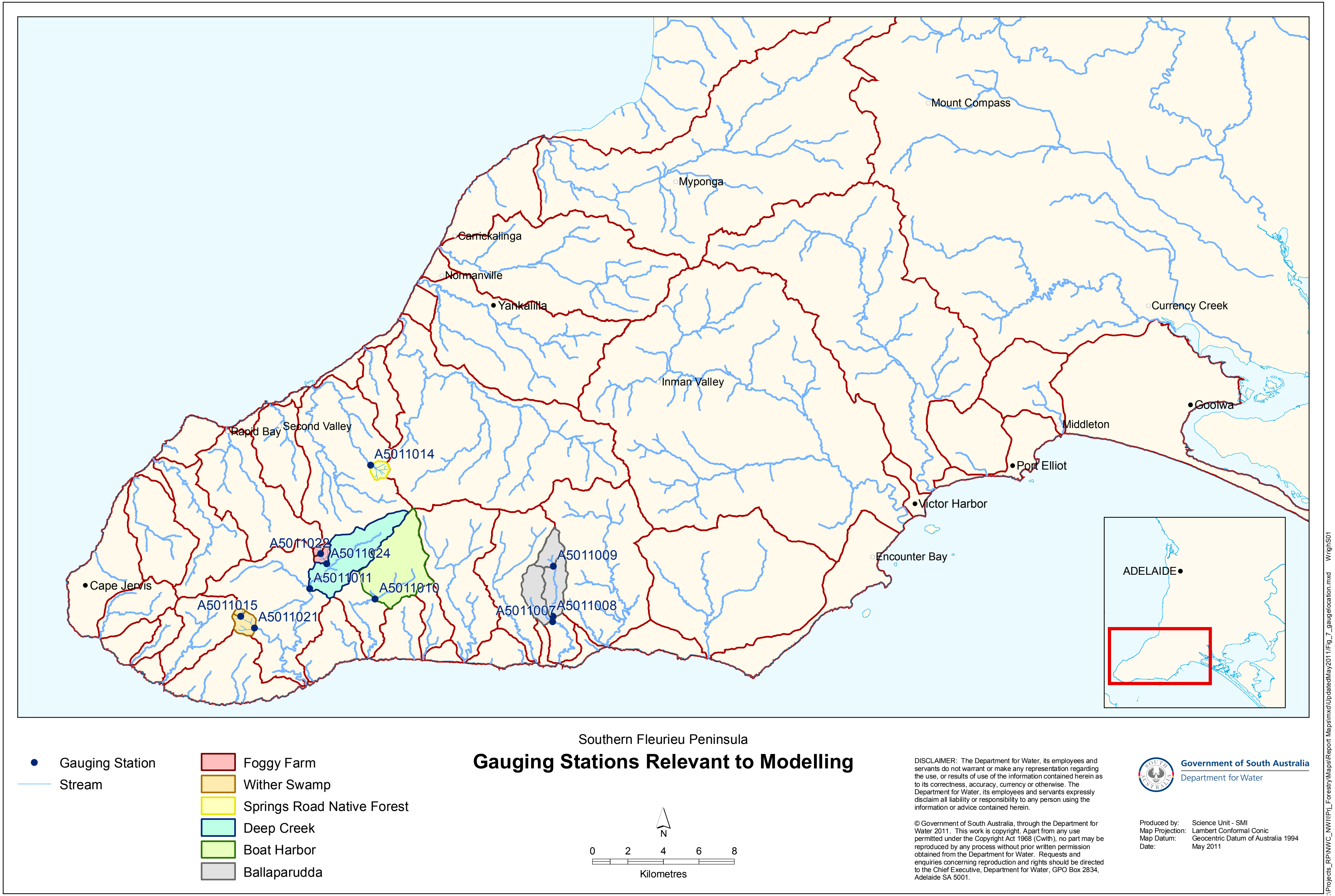


Figure 10. Location of streamflow gauging stations used in modelling

4.2. PLANT FUNCTIONAL GROUP MODELLING

The ecological analysis in this study is based upon a functional grouping of wetland plants, developed in Casanova and Brock (2000). The grouping system and theoretical water requirements for each category of plant was also recently used as a basis for estimating environmental water requirements for aquatic macrophytes in the Mount Lofty Ranges (Casanova *et al.* unpub.; Vanlaarhoven and van der Wielen 2009).

The concept of a functional group, or guild, of species is a means of simplifying natural variability. Species are classified according to their response to an environmental gradient or process of interest, with species having similar responses grouped together. By reducing the natural complexity of a biological system in this manner it allows researchers to focus more on processes driving the observed patterns. The use of this higher level of organisation was also in part a practical necessity, as insufficient field data are currently available to model water requirements at species level. It should however lead to more general and robust conclusions, as species are classified into functional groups according to relative preferences for the availability of water. This level of investigation is also more compatible with habitat management approaches required for water allocation planning.

A motivating question for this study was the likely impact of different water regimes on the observed wetland community, raising the question of what constitutes an impact. In assessing a change in the wetland plant community resulting only from human influences, a loss of any functional group would certainly comprise a loss of integrity. A permanent, measurable shift in the relative abundance of functional groups under consistent climatic conditions would be indicative of a compositional change and is also considered to represent an unacceptable impact. The latter case applies in particular to high conservation value wetlands (or species).

The plant functional groups present in Fleurieu perched wetlands include only those with a reduced need for standing water from among the full categorisation of vascular plants in the original work (Casanova and Brock 2000). Based on baseline data collected by Harding (2005) re-analysed at functional group level (Sect. 3.2), common plant functional groups (PFG) found in perched Fleurieu wetlands include : SE, Afte, Aftw, Aftl, Afrp, Tdamp and Tdry.

Excluding Aftl, which was not prevalent enough in field survey to provide adequate replication, all of these functional groups were included in the analysis. With the exception of Tdry, these functional types are capable of surviving (and in fact require) periods of both sediment inundation or saturation and exposure for varying periods (Casanova and Brock 2000, Vanlaarhoven and van der Wielen 2009). Despite being classified as a submerged group, the SE classification includes species capable of tolerating these conditions and theoretical models of the group do not require permanent water, provided sediments maintain high soil moisture levels in the root zone (Gehrig and Nicol 2010). Field survey in perched Fleurieu wetlands at the two non-permanent wetland sites used in this study (Appendix I) and in other studies (e.g. Harding 2005, Sect. 3.2), have recorded SE life forms. Clearly a subset of SE type plants found within perched Fleurieu wetlands are able to endure seasonal non-saturated conditions and are predictable plant community members in such environments (J.Nicol (SARDI) 2010, pers. comm.). On this basis the grouping was included in the analysis.

To determine what differences in hydro-period may mean for wetland vegetation communities a statistical modelling approach was undertaken. Statistical models were constructed which estimate the preferred watertable dynamics to maintain the adult life stage of selected plant functional groups from empirical evidence. Plant functional group model outputs for a given phreatic surface prediction from hydrological modelling input provides an objective basis to estimate some compositional characteristics of the resulting plant community.

4.2.1. ECOLOGICAL MODEL

Modelling is based on the premise that plant functional group (PFG) zonation will reflect individual probabilistic responses to watertable dynamics. The analysis assumes that unique (though in some cases overlapping) domains of attraction can be assigned to each functional group based on dynamics of water availability (depth, duration, rates of change, etc). For a consistent soil, nutrient, temperature and light environment, the match between plant water regime preferences (and competitive ability under different regimes) and spatial distribution of watertable dynamics will be the major structuring force for ecological expression.

While PFGs with a similar water requirement will compete, in the absence of changes to characteristic watertable dynamics, it is assumed that intra-functional group competitive succession would dominate. A species of one functional group would have the greatest probability of replacement by a superior competitor of similar water requirements—from the same functional group. In this manner proportional abundance of different plant functional groups in the absence of a hydrological change would be assumed to remain reasonably constant. The combination of relative probabilities from PFG models is assumed to be an objective indication of the likely plant community that would develop and is used as a basis for comparisons between water regimes.

The choice of descriptive variables for wetland watertable dynamics and the expected shape of the plant distribution in response to the observed range of values are critical aspects of the ecological model. A priori expectations were held for both factors. In terms of distribution, niche theory generally predicts a unimodal species response to an environmental gradient, although studies often support a skewed unimodal distribution (Austin and Smith 1989, Austin 2002). The relative position on this theoretical distribution was expected to differ for each PFG under the range of watertable variations—an important consideration in model formulation. The seasonal duration, shallow and temporary water regime observed at most of the study sites, albeit observed after an unusually dry period in this study, was considered likely to have a predictable influence on probability.

Increasing periods of inundation (hydro-period) should produce a monotonically increasing probability for submerged and high water requirement amphibious groups (SE, Afrp and higher water requirement Afte). This is illustrated in Figure 11 as the rising limb of the unimodal plant preference curve. For terrestrial groups (Tdry) the response would be expected to be monotonically decreasing. For high water requirement wetland plants this corresponds to a position on the rising linear part of the unimodal curve (from 'too dry' values towards the optimum – Fig. 11). For intermediate PFG, such as Aftw and Tdamp, preferences were expected to show first increasing and possibly optimal values (the mode of the distribution or optimal hydrological conditions – Fig. 11). If durations were sufficient, further increases in the variable will then lead to monotonically decreasing probabilities of presence (the 'too wet' values – Fig. 11). In terms of model formulation, an improved fit from inclusion of quadratic terms would signify that water requirements bracket the 'just right' value. Linear model formulation indicates functional group preferences were within the linear portion of the response curve—that is zonation was not constrained by too much water being present.

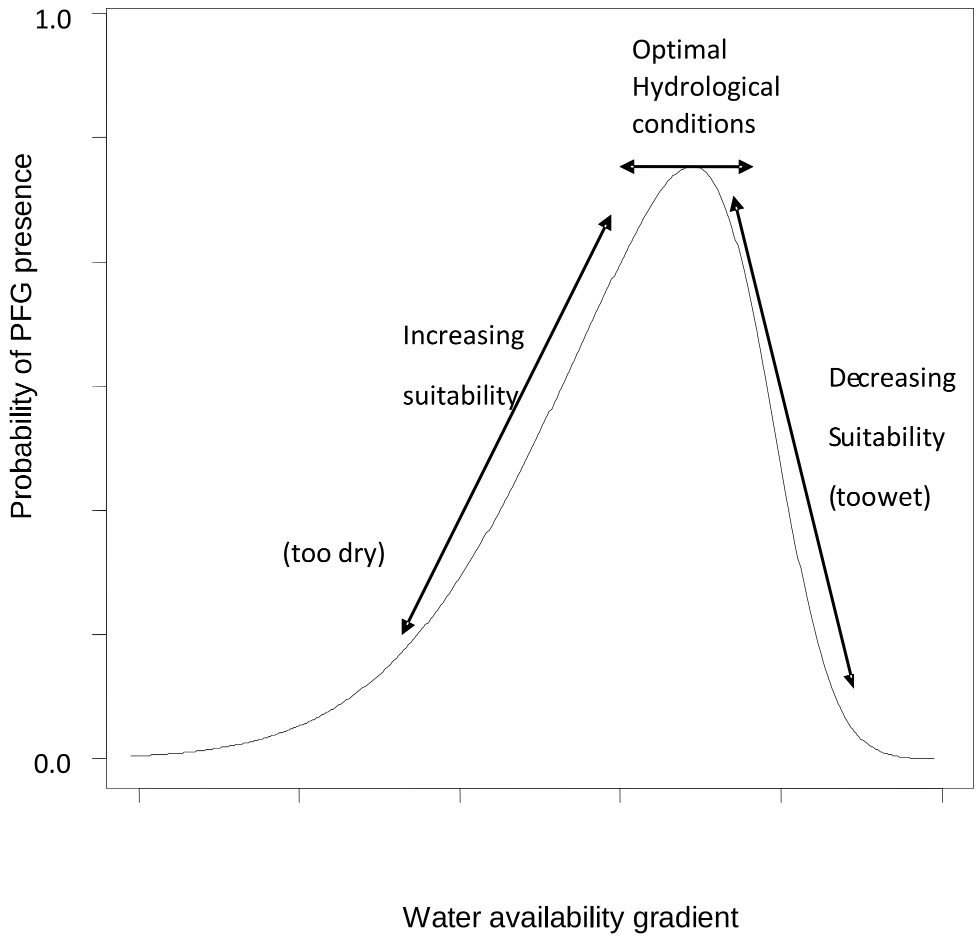


Figure 11. Theoretical functional group response curve to complex water availability gradient that exceeds that preferred for optimal growth

In terms of the best variable to describe watertable dynamics, it was anticipated that thresholds may be most easily identified at depths reflecting the root depth and distribution of each functional group. For example, having a deeper root structure, Aftw was expected to tolerate drier surface conditions than shallow rooted species. This might then lead to a preference for water regime that was most apparent in statistics representing watertable dynamics at greater depths. This assumption influenced the depths of water where statistics were calculated from continuous hydrographs, but was not used to constrain modelling variables used for each PFG. In effect, model formulation did not support this a priori speculation.

Ideally this study would have incorporated, if not focussed on, the conditions necessary for recruitment and establishment and possibly other life phases as well, which are well known as critical for determining vegetation zonation (van der Valk 1981; Rea and Ganf 1994; Van Splunder *et al.* 1995; Ganf *et al.* 2000; Keddy and Fraser 2000; van der Valk 2005; Smith and Brock 2007; Watt *et al.* 2007). Data were however not available for such an analysis and this study relies on the presence and zonation of adults as reasonable predictors of functional group water requirements. A disadvantage of this approach is that any lags in vegetation community structure may influence results. For example, where water regimes have shifted out of the preferred range, but adults are still extant in the community, this would be erroneously recorded as a viable water regime in the analysis. The duration of any lag in response would depend on the tolerance of a given functional group. For a submerged dependent group which became exposed, no effective lag will exist, while in the case of woody, relatively deep rooted groups such as tea-tree, it is likely that a lag of several years may occur. This and other study limitations will only be addressed through continual improvement of models as data becomes available from more sites over time. This work is intended to provide a starting point for such a process.

4.2.2. DATA MODEL

Data available for the study comprised hourly depth to water time series from four sites, over a 12–18 month period of record, which was linked to vegetation zones within which the relative cover of each plant functional group was estimated based on pooled species level data. Assignment of species to functional group was according to the unpublished data prepared by Casanova *et al.* for the Mount Lofty Ranges environmental water requirements project (Vanlaarhoven and van der Wielen 2009).

Depth to water data were collected using data loggers in shallow piezometers within the wetlands and co-located with vegetation transects. Water levels at each were monitored at up to four locations for periods of up to 18 months at each site. Generalised additive models (GAM) were constructed using patched point climatic datasets obtained from the Bureau of Meteorology for the nearest available site to each station. Terms used in the GAM were rainfall, evapotranspiration and a derived variable simulating soil moisture conditions based on the accumulated rainfall excess each season. The accumulated total in this dataset was reset each year to -300 mm at April 1 each year, if the accumulated rainfall deficit exceeded this value. This was done in order to ensure the values did not exceed the storage capacity of the soil and that rainfall excess each winter would accumulate from this baseline. The model fit explained more than 90% of deviance in all cases, at times exceeding 97%.

Modelled groundwater hydrographs used to construct the time series dataset represent a point estimate of watertable dynamics that can be reliably expected to be valid for a small area in the vicinity of the well. For all survey sites, digital elevation models (DEMs) were available and monitoring data were recorded at both up and downstream locations (or the equivalent positions on a gradient of water supply where more complex hydrology was observed). Together these provided the means to undertake the necessary interpolation and extrapolation to estimate water regimes at any point in the wetland. Vegetation survey point centroids were linked to the corresponding elevation on DEMs through a GIS. Elevation differences between vegetation zone centroids were used to offset the depth to water time series data by an increment reflecting the difference between the piezometer elevation and that of each vegetation survey point. Increments were modified based on observed differences in head between piezometers and interpolated (or where necessary extrapolated) based on the relative elevation differences. Spatial generalisation of this nature was not attempted for modelled scenarios and modelled wetland hydrographs represent only the wettest, most downstream point in the wetland.

The resulting ten year daily time series were analysed to provide summary statistics of the duration of the watertable at a range of depths. These were the variables used in modelling work. Variables used were median, 20th and 80th percentile depths and mean number of days per year at 0, -0.1, -0.3,

-0.5 and -1.0 m. Other statistics collected such as median periods at the same depths proved to be somewhat unpredictable and the mean days per year was the most robust of the statistics.

Sites employed were the three perched wetlands used to build the hydrological models in this study (Foggy Farm, Springs Road and Wither Swamp)—the other site being Stipiturus Conservation Park. Vegetation at Stipiturus was fortuitously mapped in 2009 (Duffield and Bailey 2010). While Stipiturus wetland is classified as Permian Sands, seasonal watertable fluctuations were similar to those observed at the perched sites. Differences naturally exist in soil texture and resultant hydraulic processes between these sites may differ. Species composition at the site reflected this and variation in functional group response according to soil type is a potential source of error in the resulting models. The depth–duration requirements for functional groups at Stipiturus were largely within the range of the perched sites, bearing in mind that the Wither Swamp site is permanently saturated and the wettest site of all. Depths and durations at Stipiturus were generally at the wetter end of the distribution of the observed seasonal perched sites, but were closer to those sites in value than the permanent sites at Wither Swamp.

Field survey data on plant cover were used to zone vegetation according to the constituent functional groups, recorded originally as species and later re-coded at functional group levels. Data were converted to presence-absence using dominance values suited to each functional group. Afte and Aftw were recorded being present where cover exceeded median values, while SE and Afrp, owing to their non-dominant biomass, were recorded present at all cover levels.

The full vegetation dataset comprises 122 samples. This was initially inspected for extreme outliers (for example, records where submerged species were recorded, but the watertable was predicted to never reach the surface over the ten year watertable dataset). On closer inspection such records were typically found to be from sites at the limits of extrapolation of watertable data and were considered unreliable indications of actual behaviour. Such records were removed. The final dataset was randomly assigned to training and testing datasets. The model building (training) dataset comprised 60% (70 samples); and, the model verification (testing) dataset comprised the remainder of these data.

4.2.3. STATISTICAL MODEL

The relationship between the binary functional group vegetation data and continuous hydro-period variables were modelled using binomial logistic regression, a form of generalised linear modelling (GLM). This approach models the log odds of a presence based on simple or multiple linear combinations of the independent variables. The underlying statistical model is that the dependent variable (the odds) is assumed to be drawn from a binomial distribution with expected value that can be modelled as a linear function of the independent variable.

Models were built after assessing correlation between the variables, which was generally high. Where variables exhibited a Pearson correlation > 0.7 only one variable was used at a time in the relevant model. Construction started with a saturated model, which included quadratic terms for PFG where theory suggests responses may be unimodal (Afte, Aftw, Afrp, Tdamp). Models were optimised by backwards selection and iteratively tested by replacing correlated variables to assess the fit for all likely descriptors. Statistics representing watertable durations at depths representing maximum likely root distribution for functional group concerned were attempted to fit first, but a greatly improved fit for all groups was observed for shallow depths. Given the nature of the wetland soils, saturation at these depths are most likely to have ecological influence. In addition 0 cm and 10 cm depth variables tended to be of lower correlation.

The usual statistical pre-requisites for model acceptance were observed, preferring simple over multiple regression predictors, linear terms over quadratic and with all parameters being significant. Significance was tested using likelihood ratios (Chi square distributed with degrees of freedom equal to model parameters). Chi square tests were also used to test nested models for significance of the additional parameter. Final models adopted were those that maximised the explained deviation provided these met other modelling criteria. Models reflecting ecological theory around plant functional group water requirements were preferred. This limited candidate models to those exhibiting either a monotonically increasing (submerged emergent) or unimodal (amphibious groups) response to watertable dynamics. Models were also rejected unless representing a statistically significant improvement on that of chance at the 99% level for the test dataset. The need for robust estimates and higher generality was favoured in the choice of final models and a number of multi-parameter models were rejected on parsimony or lack of consistency with ecological theory, despite excellent classification performance and explaining larger proportional deviance.

The formula and parameters for the models are presented in Appendix C, with selected model verification statistics (Fielding and Bell 1997) shown in Table 2. Shallow depth to water dynamics were found to be especially important, with surface and near surface durations the best predictors. Unfortunately this is also the main uncertainty of the study as near and particularly at surface, watertable dynamics are difficult to extrapolate across elevation gradients with certainty in order to acquire sufficient vegetation data for a statistical approach. This difficulty was also observed in the hydrological modelling, which while representing general watertable behaviour well, was not capable of reproducing the subtleties of near surface dynamics.

Table 2. Binomial logistic regression model verification statistics

Plant functional group	PseudoR2[(1)]	AIC[(2)]	Correct classification rate	Positive predictive power	Negative predictive power	Pr(Z > z) [(3)]
Afte	44%	63.6	0.76	0.88	0.69	1.50E-04
Aftw	33%	67.3	0.76	0.72	0.81	4.83E-04
Afrp	24%	73.9	0.69	0.82	0.62	2.55E-03
SE	27%	64.6	0.91	0.78	0.97	5.64E-06
Tdry	39%	59.4	0.89	0.95	0.85	3.82E-08
Tdamp	31%	67.6	0.72	0.73	0.72	1.13E-03

1. Proportion the null deviance explained in the model.
2. Akaike information criterion – a measure of model fit
3. Probability of obtaining the classifications observed through chance alone

Deviance explained varied from 28–47% of total deviance and models attained correct overall classification rates between 0.72 to 0.89 (Table 2). Although both statistics are relatively modest, these were considered robust and adequate for the purposes of this study. In general negative predictive power was higher than positive predictive power, meaning models predict absence with higher confidence than presence. All models represented a statistically significant classification capability ($p < 0.01$) over that which could be expected from chance alone and provide support for the general approach of using hydro-period to predict plant functional group presence.

4.2.4. ANALYSIS OF FUNCTIONAL GROUP MODELLING

The plant functional group models were run with the summary statistics for watertable dynamics dataset from each of the modelled land use scenarios as well as the observed and modelled site data. The output from the models is predicted probabilities for the presence of each PFG, based on the fitted model and predicted watertable dynamics within each scenario. Owing to the low certainty in watertable dynamics near the surface, it remains uncertain as to how close these may be to actual absolute probabilities. These uncertainties are common to all scenarios however and findings can be interpreted with more confidence in a relative sense, which was the essence of the study.

Probabilities for each functional group and scenarios of highest interest to this study were used to create measures of association (dissimilarity matrices) between scenario and study site water regimes. This provided an objective comparison of the likely similarities between the resulting vegetation communities. As the joint absence of a functional group from two scenarios is explicable under the theoretical underpinnings of the model, this carries valid ecological information. As a result both Euclidean and the more typical ecological Bray Curtis (Bray and Curtis 1956) measures of association were used to create dissimilarity matrices of the PFG probabilities using the statistical programming language R (R Development Core Team 2010), with functions included in the package 'vegan' (Oksanen *et al.* 2010). Resulting matrices were then subjected to the agglomerative hierarchical clustering method UPGMA (Unweighted Pair Group Method with Arithmetic Mean), implemented in R through the 'agnes' function within the 'cluster' package (Maechler *et al.* 2005). The results of the two different distance measures in further analyses were extremely close and the results of the Bray Curtis clustering are presented. Dendrograms were pruned to create groupings tested for statistical significance using permutational multivariate analysis of variance (function 'adonis' in package 'vegan' Oksanen *et al.* 2010). GAMs used to model depth to water at field sites were constructed in R using package 'mgcv' (Wood 1994).

A non-metric multidimensional scaling ordination of sites proved to be non-informative and is not presented. Owing to the linear dependence of functional group probabilities on a single category of phreatic surface variables, the majority of variation within the dataset projected in multi-dimensional space was directed along a single axis which was highly correlated with wetland hydro-period.

4.3. HYDROLOGICAL MODEL CONSTRUCTION AND EVALUATION

4.3.1. MODELLING OVERVIEW

Hydrologic models are conceptual models that represent the behaviour of various components of the hydrological cycle and also how those various components interact. Rainfall, interception, evaporation, infiltration, surface runoff, surface–groundwater interaction and baseflow are some of those key components. The components and the interaction between those components are represented by mathematical functions that are built into a model by using computer-programming languages. The models are built to simulate catchment conditions and to generate long term hydrological data from observed historical data. These models enhance our understanding of hydrological behaviour of the catchments and are further used for the assessment of impact of various changes and activities within the catchment.

For this study, hydrologic modelling of a catchment involves the following three-step process:

Model construction – the representation and spatial distribution of the physical features and processes that influence the generation, movement, diversion and capture of runoff in a catchment. This is undertaken by creating a series of interconnected nodes, each corresponding to a different feature within the catchment. Each node is characterised by a series of mathematical equations which represent the various processes and their relationships in the hydrological cycle.

Model calibration – simulation of catchment conditions in the model to generate runoff to match the actual observed runoff is generally termed as model calibration. An iterative process is the only method available to solve the transfer equation as direct physical measurement of the parameters is difficult. This was undertaken by manual manipulation of parameter values till a suitable correlation between computed and observed hydrographs was achieved. Ideally, the same set of calibrated model parameters can be used for subsequent simulation with little or no further adjustments.

The level of efficiency of the calibration process depends on availability and accuracy of the number of hydrological parameter datasets. Since the hydrological cycle involves a large number of parameters that are not measured, efficient calibration of hydrological models require good knowledge of catchment conditions, in addition to input datasets.

Modelling scenarios – this is the process of running the calibrated model with a long term hydrological dataset(s) to obtain long term estimates of other hydrological dataset(s) that were not measured to:

- provide a historical insight of hydrological condition of the catchment
- assess the probable impacts of various changes that have occurred in the past on catchment hydrology
- assess the impacts of possible future development and changes on catchment hydrology.

4.3.2. MODELLING PLATFORM

WaterCRESS (Version2) (Cresswell, 2002), a PC based water balance modelling platform was used for construction of the model in this study. This modelling platform incorporates some of the most widely used models in Australia. WaterCRESS allows the incorporation of different components like catchment component, water use component etc. to represent generation, use, storage and transfer of water in its water balance models. WaterCRESS incorporates a choice of rainfall–runoff models and WC-1 has been commonly used to construct and calibrate models for various catchments in South Australia and hence was used in this study. A detailed description of the WC-1 rainfall–runoff model is in Appendix J.

4.3.3. MODEL CONSTRUCTION

Within WaterCRESS (WC), model construction is undertaken by creating a series of interconnected nodes, each corresponding to a different feature within the catchment. Each node is characterised by a series of mathematical equations which represent the various processes and their relationships in the hydrological cycle.

WC models were set up for three catchments: Foggy Farm, Wither Swamp and Springs Road native forest. A model was also constructed for one additional catchment, Burnt Out Creek, in order to model rainfall–runoff over the pine plantation establishment phase.

Each model catchment was subdivided into separate wetland and contributing catchment components. Wetlands were considered a partial catchment area that would contribute flow to a stream via saturation excess overland flow and subsurface flow. The contributing catchments are upslope areas contributing flow to the wetland area.

4.3.3.1. Model nodes

Each contributing catchment is represented via a rural catchment node, while the wetland is represented using a reservoir node (storage node), which gets filled up before draining out. The catchment parameters describe the rural catchment node and the Flow-Elevation-Volume-Area (FEVA) file is used to describe the storage characteristics in the reservoir node. Each of these nodes is linked with a drainage line showing flow in and out of the component.

The inputs to each node are described below.

Catchment node inputs:

- Area of catchment draining, representing that node
- Corresponding observed daily rainfall dataset , daily evaporation data
- Catchment parameters (listed in Table 4) to the rainfall-runoff model used (in this case WC-1)
- Calibration file, which contains corresponding observed stream flow dataset for the node that has gauging station.

Reservoir and dam node inputs:

- Capacity
- Corresponding observed daily rainfall dataset, daily evaporation data
- FEVA file representing flow elevation volume and area relationship in reservoir
- Calibration file, which contains corresponding, observed stream flow dataset for the node that has a gauging station.

4.3.3.2. Modelling Steps

The modelling steps are illustrated in a flow chart (Fig. 12) and described as below.

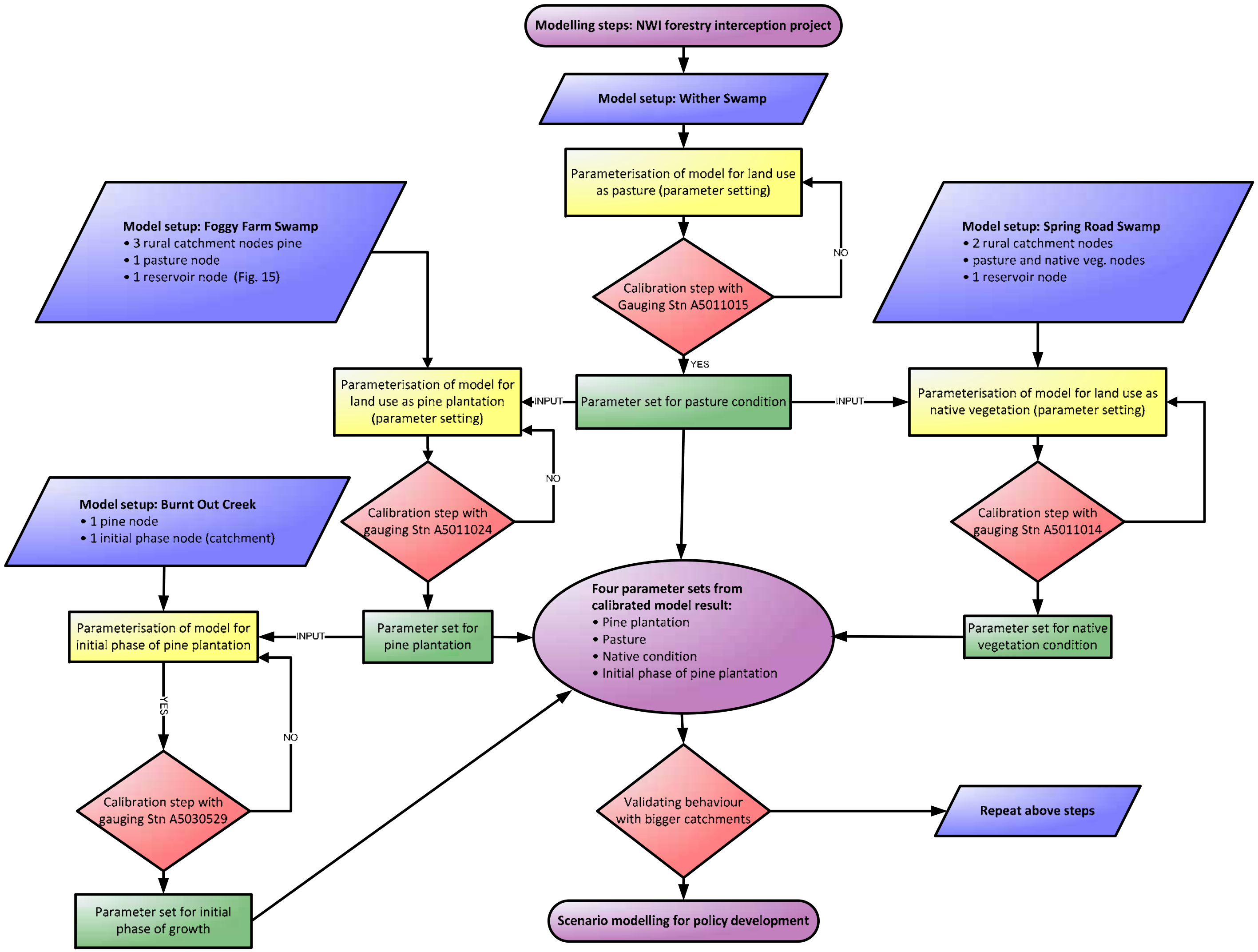


Figure 12. Steps involved in modelling

4.3.3.3. Model setup and calibration

The model setup for each catchment and its calibration is described in sequential order in this section. As illustrated in Figure 12, the first model was setup with a single rural node for Wither Swamp. The catchment area was 18.6 ha with uniform land use of pasture. Rainfall data from the local pluviometer (station no: A5011018) data were used in the model while the monthly evaporation data were taken from the neighbouring station M023783 (Myponga reservoir)

This model was calibrated against the recorded flow for the year 2009 from station A5011015 (Fig. 13) to get a parameter set representing the hydrological response for the pasture land use. The calibrated parameter was then used as the input to the model setup of Foggy Farm and Springs Road native forest catchments in order to determine unique parameter sets for the respective land covers.

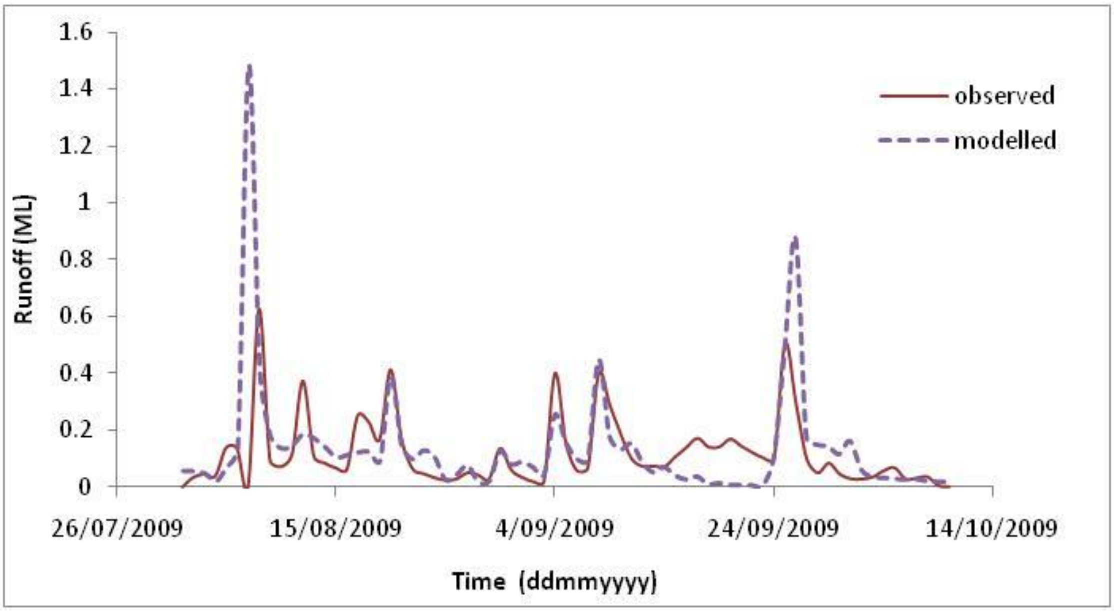


Figure 13. Modelled and observed daily flows at u/s Wither Swamp gauging station A5011015

The second model setup was for the Foggy Farm catchment. It had two types of land use: pasture and pine plantation. For this model setup (Fig. 14), four rural catchment nodes (three representing catchment areas with pine plantation with different catchment areas and one representing the portion of catchment with pasture) were used. The pine catchment (node: pine2; 35.9 ha) drained into a pasture rural catchment (node: past1; 1.3 ha), which then drained to the wetland (node: wet1). The upstream catchment (node: pineus 28.3 ha) was the catchment for the dam (node: dam1). The spill from node dam1 flow into the reservoir node (wet1). The last catchment (node: pine 26.6 ha) drained into the reservoir node (wet1). The swamp area was represented by the reservoir node (wet1).

The surface area to volume relations for the wetland were represented by the FEVA file in the model. The rainfall data was taken from Deep Creek Upstream Foggy Farm Dam spillway (station no: A5011022). The farm dam was represented as an on-stream dam node (dam1), from where water was being pumped out. The volume pumped out from the farm dam was recorded in the monitoring site A5011022.

The model was then calibrated against the flow recorded at the downstream end of Foggy Farm (station no: A5011024) as shown in Figure 15 for 2009. The catchment parameter set for the calibrated model represented the catchment parameter set for pine plantation.

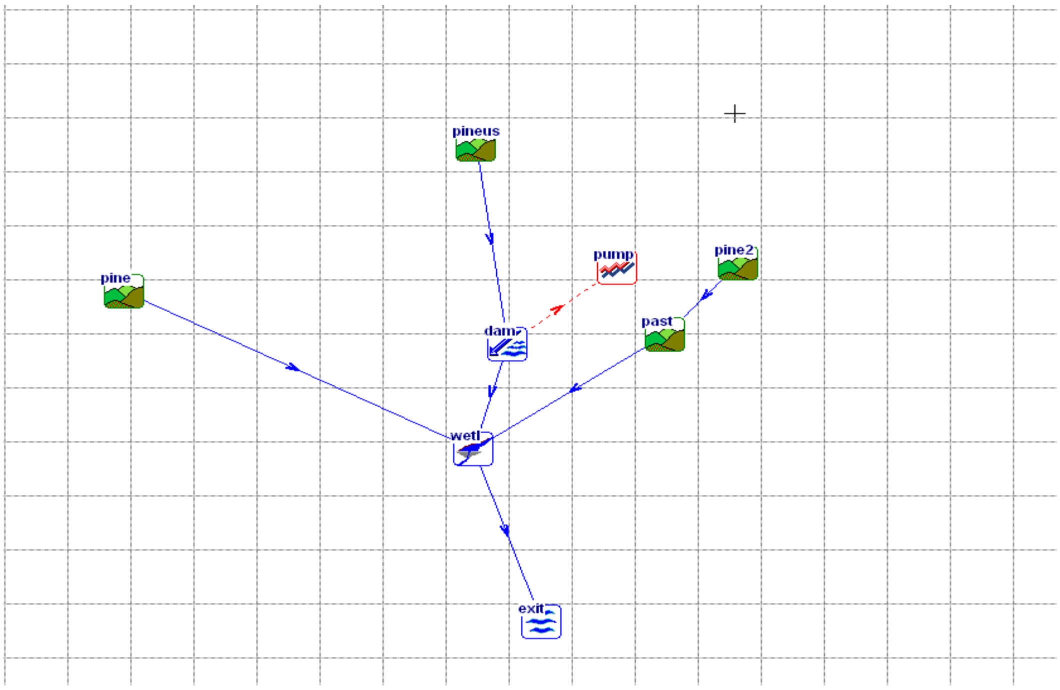


Figure 14. Foggy Farm model setup

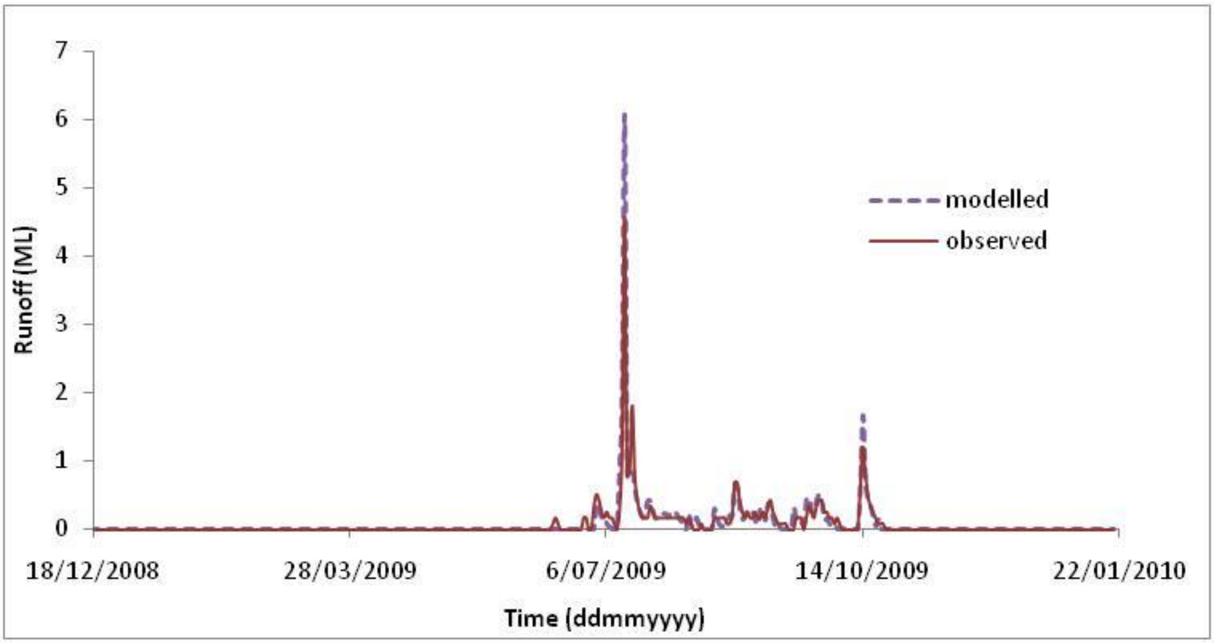


Figure 15. Observed and modelled daily flows at gauging station A5011024 downstream Foggy Farm

With the calibrated parameter set of pasture, the model was setup for Springs Road Native Forest Reserve. The model had two rural nodes, one representing pasture land use of 15.3 ha and the other representing native vegetation of 72.2 ha, draining to a wetland node

The storage of wetland was represented by the FEVA file in the model. The rainfall data for this site were from a pluviometer within the catchment (station: A5011025).

To generate a parameter set for a catchment under native vegetation the model was calibrated against the flow recorded from the gauging station at Springs Rd native forest (station: A5011014). Calibration results for native vegetation are shown in Figure 16.

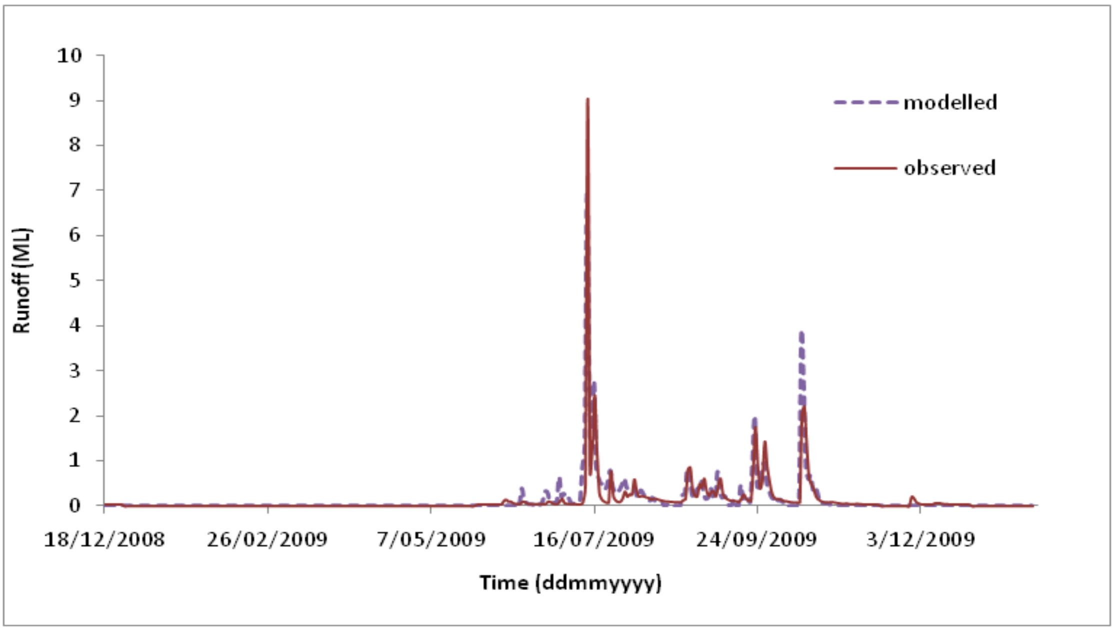


Figure 16. **Observed and modelled daily flows at gauging station A5011014 downstream Springs Road Native Forest Reserve.**

During the first few years of pine plantation establishment, vegetation cover is extremely limited. Use of a model calibrated to the closed canopy response would underestimate runoff and a parameter set was required for the initial years of the rotation cycle to simulate this phase of forest development. Rainfall and streamflow data from Burnt Out Creek over the period 1978–82 reflected such an establishment phase. By building a model of the catchment and calibrating to data over this period a parameter set indicative of catchment runoff during the establishment phase was obtained.

The 60 ha Burnt Out Creek catchment was modelled with two rural catchment nodes representing pine catchments: one (40 ha) representing the initial phase of plantation and another node with calibrated parameters for pine plantation. The model was then calibrated against gauging station A5030529 (Burnt Out Creek Upstream of Mount Bold Reservoir) to have a calibrated parameter set for the initial phase of pine plantation. Figure 17 shows the calibration at Burnt Out Creek from the period 1978–82. (Greenwood and Cresswell, 2007).

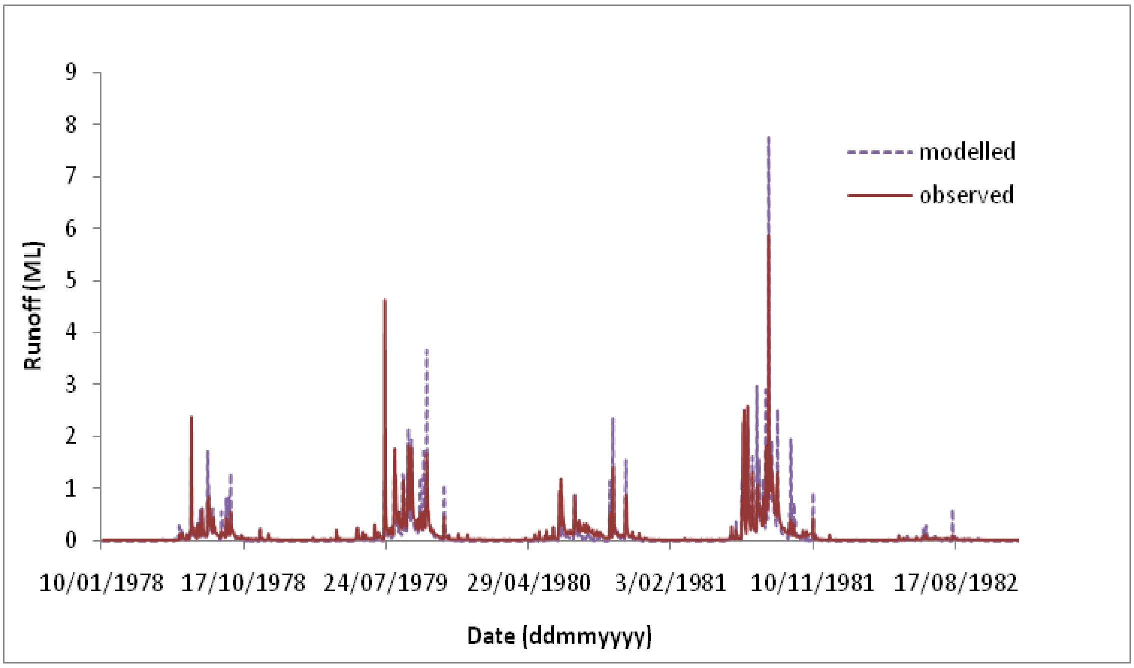


Figure 17. Observed and modelled daily flows at gauging station A5030529 Burnt Out Creek

The calibration figures show the observed and daily modelled flows for the four catchments: Wither Swamp, Foggy Farm, Springs Road Native Forest Reserve and Burnt Out Creek catchment (Figs 13 and 15–17 respectively). Calibration statistics indicate a good correlation between the observed and modelled flow (Table 3). However, it should be noted that the calibration has been done with only one year's streamflow data for the catchments, excepting Burnt Out Creek. Long-term data is required to give more confidence in model calibration.

Table 3. Calibration statistics for the four models used to develop scenarios

Model	r-squared	Coefficient of efficiency	% Volume difference
Wither Swamp model	0.88	0.62	2.59
Foggy Farm model	0.95	0.84	0.96
Springs Road model	0.91	0.84	0.604
Burnt Out Creek	0.85	0.72	0.148

Parameter values for the WC-1 for the four catchments representing the different land uses are presented in Table 4. Note that the two parameters conceptually representative of catchment soil moisture characteristics are held constant across all scenarios (MSM and CD). This was intentional and done to ensure land use response was the primary difference between scenarios.

Table 4. Parameter set for different land use (see Appendix J for parameter names and definitions)

Parameters of WC model	Land use			
	Initial phase	*Pasture*	*Native vegetation*	*Pine plantation*
MSM	240	240	240	240
IS	12	16	27	37
CD	80	80	80	80
GWD	0.1	0.1	0.1	0.09
SMD	0.00029	0.0005	0.00025	0.0001
PF	0.78	0.85	0.9	1
FGL	0.001	0.001	0.001	0.001
SWM	0.88	0.85	0.85	0.85
GWR	0.42	0.4	0.36	0.36
CL	0	0	0	0
KS	0	0	0	0
CI	0	0	0	0

4.3.3.4. General behaviour

The parameters generated from the calibrated results were then applied to larger catchments in the region for validation of the general behaviour of catchments and determine any scale related issues. The bigger catchments used were Boat Harbor (1274 ha) and Upper Deep Creek (1042 ha). These catchments were selected because of the similar hydrological and land use characteristics of the catchment and close proximity to the research catchments. The results were compared with flow data from gauging station A5011011 for the upper catchment of Upper Deep Creek (Fig. 18) and station A5011010 for Boat Harbor (Fig. 19) catchments. The general behaviour was replicated in the model with the same set of parameters from calibrated model, with over prediction of volume difference up to about 14%.

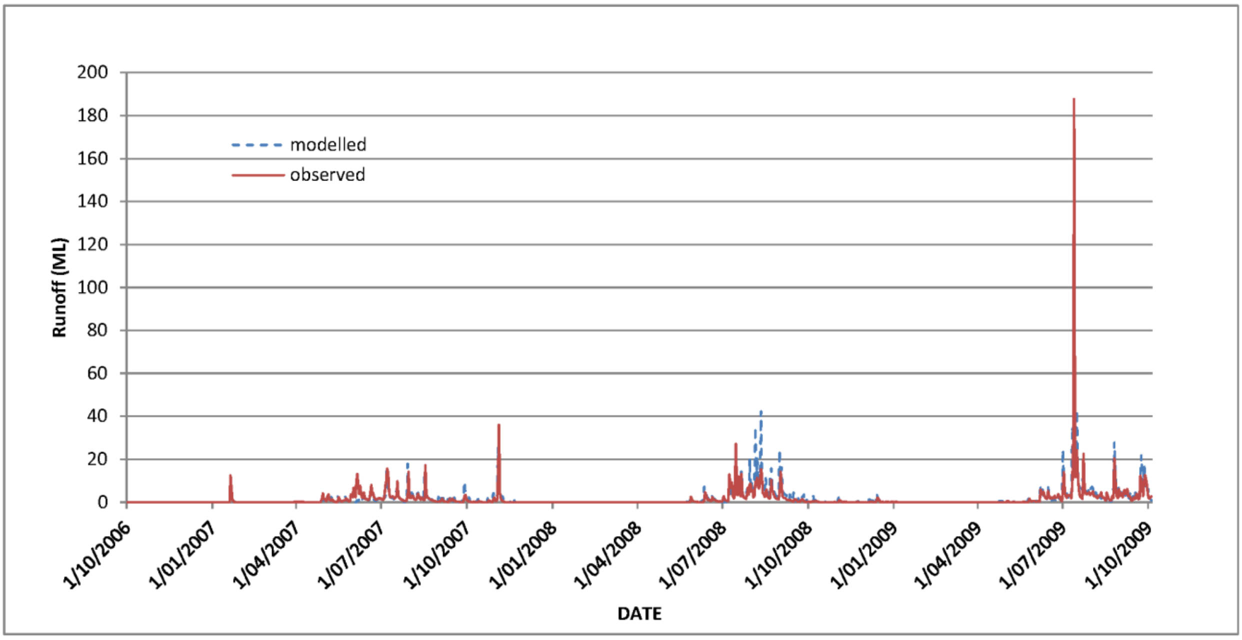


Figure 18. Modelled and observed daily flows at gauging station A5011011 (Upper Deep Creek) for general behaviour

These catchments were selected because of the similar hydrological characteristics of the catchment, their proximity to the research catchments and their longer period of record. It is stressed that no attempt was made to adjust model parameter sets, only areas. The exercise provides a cross-scale test of general model performance in the region, to determine whether the small scale of modelling has produced unreasonable runoff expectations. This was considered a good response, as no farm dams were incorporated in the models and rainfall used was a single estimate from Parawa, an unrealistically high estimate for lower catchment areas. The results of the validation exercise indicate the calibrated model parameter sets provide robust estimates of runoff for the region.

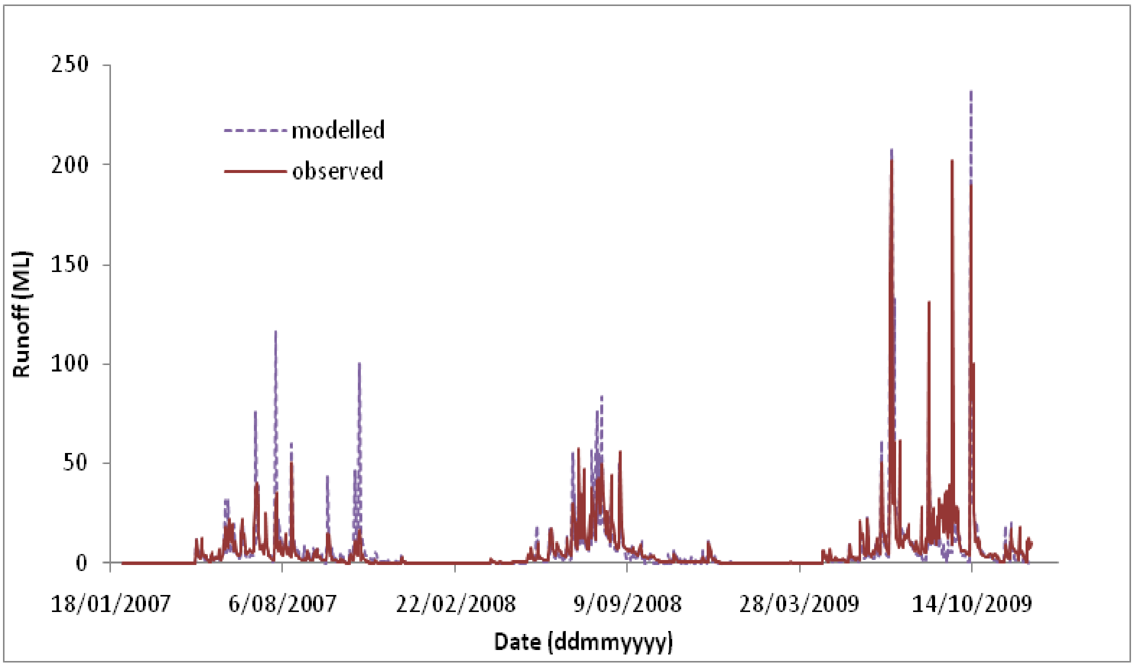


Figure 19. **Modelled and observed daily flows at gauging station A5011010 (Boat Harbor) for general behaviour**

4.3.4. WETLAND GROUNDWATER STORAGE DYNAMICS

Wetland volumetric storage was estimated based on observed physical dimensions (wetland length, width, depth) and porosity estimated from measured hydraulic properties (Appendix F) and watertable response to rainfall (Sect. 5.2). The depth at which stream flow commenced was estimated based on the relationship between observed streamflow and logged groundwater level data at an adjacent piezometer (Sect. 5.1 & 5.2). Dimensions of the clay subsoil store were 2.2 m saturated depth secondary porosity of 0.1 and area of 2.34 ha giving 5.15 ML storage. These dimensions were linearly interpolated to obtain the flow-elevation-volume-area (FEVA) file used to model the wetland storage. The surficial loam layer had modelled dimensions of 30 cm thickness, porosity of 0.65 and area 2.34 ha representing 4.6 ML of storage. As the relationship between modelled storage held and depth estimates for each layer were effectively linear, this allowed wetland groundwater dynamics as a whole to be simulated using piecewise linear regression of depth to water on storage held in the two layers at a daily timestep.

Storage volume input to the linear equation relating to the clay layer came directly from the rainfall–runoff models, which generated daily time series data for the wetland (reservoir node Sect. 4.3). This represents the volume held within the residual storage of the wetland (lower clay layer). The residual storage of the wetland represented in the model as the FEVA file coincided with the top of the clay layer in the conceptual model.

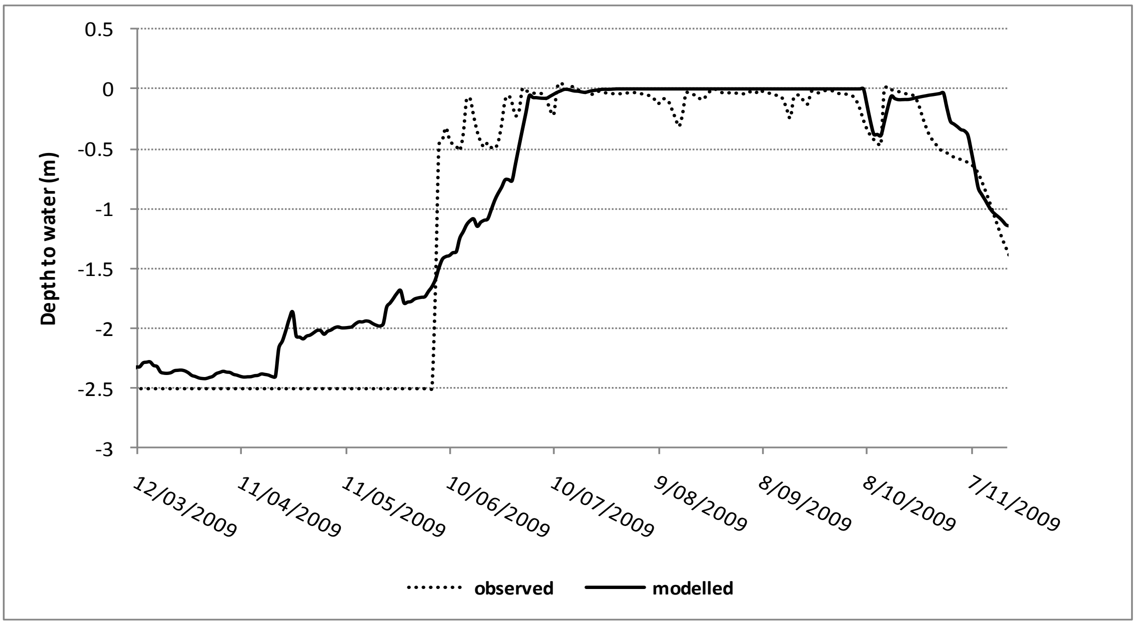


Figure 20. Observed and modelled groundwater hydrographs – Foggy Farm catchment outlet

General wetland storage dynamics were fairly well represented (Fig. 20), but the modelled watertable remains constantly at the surface in the model, while observed levels drop below ground and return to the surface in response to rainfall events. Watertables in streambed sediments adjacent to the surface water flow gauge remained at the surface more consistently than within the wetland (see Fig. 20), as the area receives flow from the entire catchment and this is concentrated within the watercourse channel (see Appendix H for instrument locations). Days per year that the storage was full and water tables were at the surface are likely to be somewhat over-estimated in the model as a result. While this affects the absolute probabilities for the vegetation modelling where logistic equations use the days per year at the surface, the systematic over-estimate applies across all scenarios and the relative vegetation probabilities are considered to be the main finding of interest (discussed in Sect. 4.2). Model outputs from each scenario were input using the same water balance approach to estimate the relative changes in the wetland storage and were used as the basis for vegetation modelling.

Of note in Figure 20 (see also Fig. 25) is the hydrograph response in the observed data. The vertical section of the curve in early June suggests an overnight rise of almost two metres. This is not considered to be an accurate reflection of overall dynamics however due to the nature of the wetland sediments. Cracks transmitting water in the subsoil were not intercepted by the piezometer and water did not flow into the well until the wetland subsoil was filled and saturation of the surface organic layers occurred (Sect. 5.2). As such, the modelled data likely presents a more representative, though slightly delayed, hydrograph of the early season filling of the wetland residual storage. Consistent streamflow commenced in early July at the site (Fig. 25).

4.4. MODELLING SCENARIOS

4.4.1. BENCHMARK USED FOR COMPARISON OF SCENARIOS

In order to assess the effects of different scenarios on hydrology and ecology, a point of comparison was required. Maximum catchment yields are associated with pasture land use and this was adopted as the benchmark used to assess changes in this report. In reality ~1.9 ha (or 10%) of the 18.6 ha catchment was under deep rooted perennials and this probably represents a fairly typical ratio of tree cover for cleared paddocks with shelter trees planted along paddock boundaries in the region.

4.4.2. DESCRIPTION OF VARIABLES AND SCENARIOS

Section 1.1.1 describes the criteria that are used to determine forest planting configurations in South Australia for environmental and water resource protection. The two criteria, allowable catchment planting proportion and required buffer width, were manipulated and different combinations of these were run as individual scenarios to estimate possible direct and interaction, effects of these. Table 5 lists the different values adopted for the two variables under investigation and indicates the analysis undertaken to determine the resulting implications.

Table 5. Scenarios used in modelling

Criteria under test	Scenarios run	Analysis of results
10%, 25% rule*	Five different planting proportions: 12, 30, 50, 70 and 100%	Annual and monthly runoff statistics; Predictive modelling of dominant vegetation functional group
Buffer distances	Four different buffers sizes: 10, 20, 50 and 100 metres	(as above)

* Statewide forestry policy and draft water allocation plans – note the use of the pasture scenario as the benchmark for comparison in this report is differentiated from the mixed land use benchmark that applies to state policy.

Scenarios run were assigned a numerical code to identify the assigned buffer distance and plantation proportion (Table 6).

Table 6. Different scenarios modelled (based on a 100 ha catchment)

Buffer zone / *Planting fraction*	10 m buffer *(1.4 ha)*	20 m buffer *(2.8 ha)*	50 m buffer *(7.4 ha)*	100 m buffer *(16.3 ha)*
12%	Scenario 1	Scenario 2	Scenario 3	Scenario 4
30%	Scenario 5	Scenario 6	Scenario 7	Scenario 8
50%	Scenario 9	Scenario 10	Scenario 11	Scenario 12
70%	Scenario 13	Scenario 14	Scenario 15	Scenario 16
100%	Scenario 17	Scenario 18	Scenario 19	Scenario 20

All scenarios are based on a 100 ha catchment contributing area, including a wetland of 6.3 ha receiving inflow from the catchment via the buffer area corresponding to the modelled buffer distance. The dimensions of the wetland are based on estimates derived from the Foggy Farm and Springs Road wetlands in the study catchments as described in Section 4.3. The resulting ratio of wetland to catchment area was based on an average of field sites in this study, but this may be lower than typically observed. A random sample of perched wetlands data had a mean proportion of wetland to catchment area over 10%, while in this study the wetland was ~7% of the total modelled catchment area. Larger wetlands constitute a greater demand and will likely respond differently to hypothetical scenarios in this study.

All scenarios were run with 37 years of data (1973–2009), at a daily time step, with climate data extracted from the Bureau of Meteorology SILO PPD (patched point datasets) for Parawa (Sharon) station M023761. The long term rainfall and evaporation data used are the same for all scenarios and considered representative of the area around the catchments under study.

Runoff responses were analysed at annual and monthly time steps. Summary statistics were calculated to provide an indication of changes between responses under pasture, native vegetation and the various pine plantation scenarios.

5. RESULTS AND DISCUSSION

5.1. *FIELD DATA*

Summary surface and groundwater data are presented in this section. Soil description and hydraulic characteristics are presented in Appendixes D and F respectively. Vegetation survey data and analysis are found in Appendix H and selected volumetric soil moisture data and interpretation appear as Appendix I.

5.1.1. SURFACE WATER DATA

Stations commenced monitoring in spring 2008, a year of substantial drought conditions across the state and streamflow had ceased by the time sites became operational in September at Springs Road and Foggy Farm. As a result, no streamflow data were collected in 2008 from these sites and only the 2009 data are presented. Summary data from the three field sites are presented below for the 2009 monitoring year (Table 7).

Table 7. Hydrological summary statistics for 2009 at field sites

Site:	Pine *(Foggy Farm)*	Native forest *(Springs Road)*	Pasture *(Wither Swamp)*
Rainfall (mm)	963	747	831
Streamflow (ML)	32.0	53.7	380
Area (ha)	105	88	151
Depth of runoff (mm)	39	61.0	253
Runoff coefficient	4%	8%	31%

Rainfall varied by over 200 mm between the driest and wettest sites (Table 7), indicating the gradient from south to north across the Parawa plateau. Springs Road is the only site to the north and received the lowest rainfall. Streamflow was the most variable of the results: runoff coefficients indicate Foggy Farm generated around half the runoff of the native forest catchment, while the depth of runoff from the Wither Swamp exceeded 30% of rainfall. In contrast, the runoff coefficient for the upstream gauge at the site for 2009 was 14%. The major difference between the upstream and downstream areas at the site is the influence of the local groundwater flow system (see Sect. 5.2.), which does not affect upstream runoff.

Daily time series streamflow data for the three sites are presented in Figure 21. Peaks in the winter–spring flow season are associated with high rainfall events and generally coincide across the three sites. The response from the pasture site features perennial baseflow and much higher relative responses to each of the rainfall events. The influence of early season rainfall is also evident in the pasture catchment, with flow events occurring consistently from April through until November. The two catchments with seasonal flow have comparable flow durations, largely from July through to October. Generally flow peaks are higher throughout the season at the native vegetation site than the plantation site. The greatest uncertainty in streamflow data relates to the native vegetation study catchment, which also included an area of around 15% of the catchment under pasture at the upstream end of the wetland. As this input could not be gauged directly to determine the precise level of runoff from the native woodland, it was necessary to estimate this contribution through modelling. Anecdotal observation subsequent to the monitoring period suggests the pasture response may have been overestimated, which would have resulted in an unrealistically low runoff from the woodland area (see Sect. 6.3.3.1).

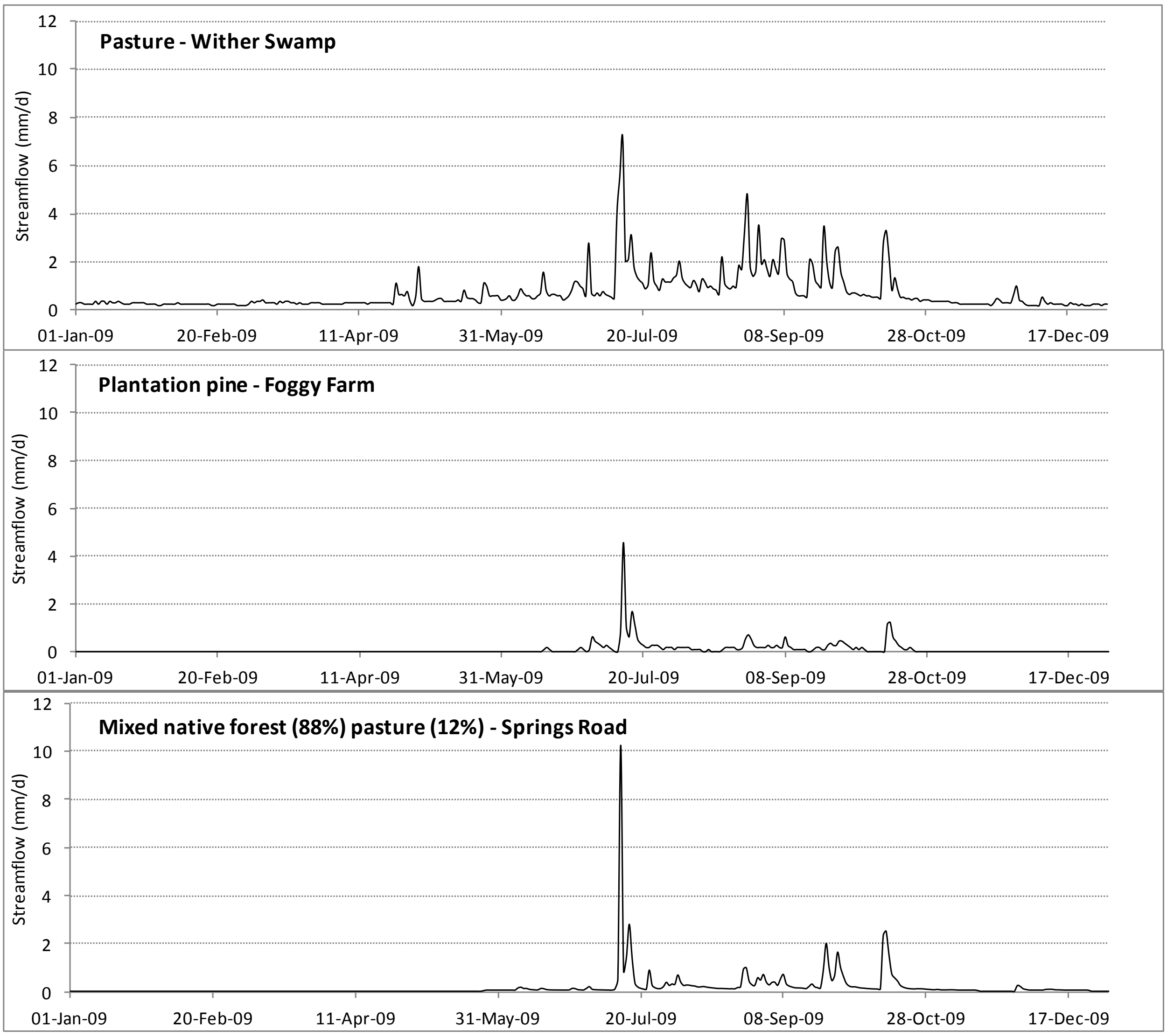
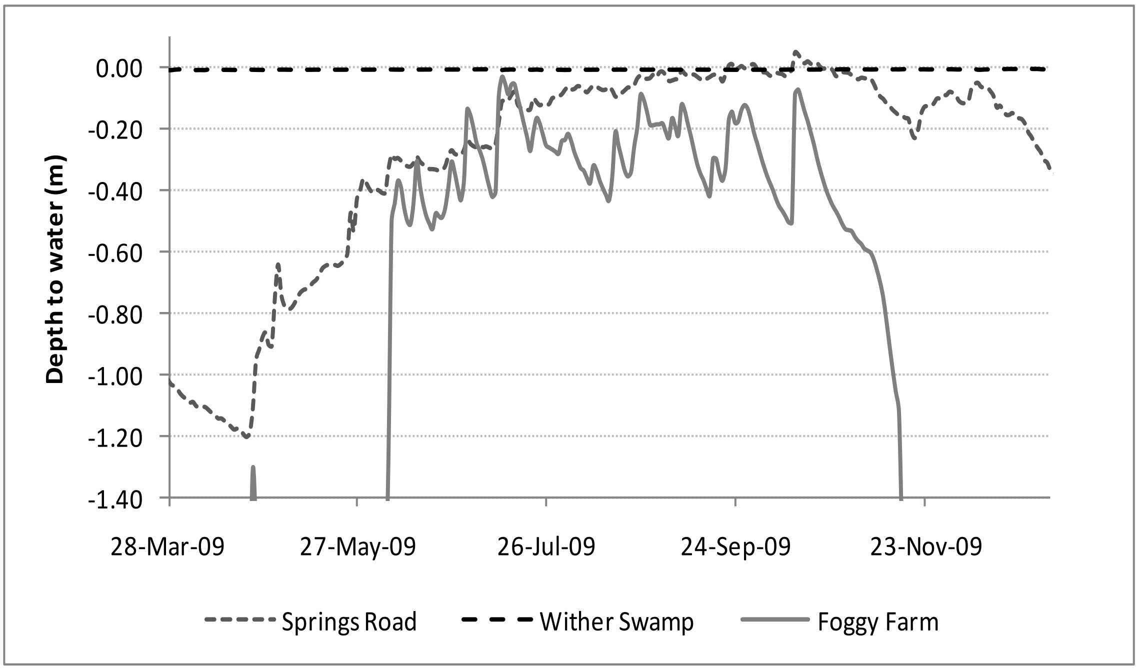


Figure 21. **Streamflow from the three downstream gauges for 2009**

5.1.2. GROUNDWATER DATA

Figure 22 demonstrates the wide variation observed in wetland water table dynamics at the three field sites. All hydrographs are from observation wells at locations believed to represent the wettest areas of each wetland. The Foggy Farm hydrograph comes from the central tea-tree zone (FF03). This hydrograph contrasts with the one shown in Fig. 20, which is from a location adjacent to the streamflow gauge, data from which were used to calibrate the surface water model reservoir node (Sect. 4.3). The hydrograph presented for Springs Road is from near the streamflow gauge, so as to avoid any influence from runoff entering the wetland from the upstream pasture area (Appendix G) and is considered to represent wetland ambient water levels. These differences in hydrology were reflected in the resulting vegetation communities, with water availability a major structuring factor (Appendix H). Hypothesised causes for the different dynamics are discussed in Section 5.2.



Note the vertical rise in levels in the Foggy Farm hydrograph – as explained in Section 4.3 this is due to the characteristics of the wetland subsoil water movement via preferential pathways such as inter-aggregate spaces and macropores.

Figure 22. Daily groundwater hydrographs from the three field sites Apr–Dec 2009

5.2. CONCEPTUALISATION OF PERCHED WETLAND HYDROLOGY

5.2.1. CATCHMENT AND WETLAND SOIL CHARACTERISTICS

Details of soil survey and hydraulic characterisation are presented in Appendixes D and F. Surveys indicate that catchment soils have formed largely in-situ from underlying metasediments (mostly meta-siltstone) and textures reflect the vertical progress of the weathering front.

Most soils occur on hillslopes between plateau surfaces and wetlands. These typically have loamy textured surface soils that grade to or overlie light to medium clay, which in turn grades to highly weathered rock (usually at around 1 m) and ultimately to fresh parent material. Topsoils are generally 10–40 cm thick. Many soil layers are rich in silt-size particles and so have silty textural grades (e.g. silty light clay); this is especially the case in lower subsoil layers, but occurs throughout many profiles, indicating soil profile connection with silt-rich parent material, but also indicating relatively young soil developmental age (as the mica-rich silt has not yet weathered to clay-size particles), which is unsurprising on slopes in these naturally erosional landscapes. The youngest and shallowest soils have no clayey subsoil, but have clay loamy transition layers that grade directly into the underlying weathered rock. Soils on lower slopes are characterised by the presence of a bleached subsurface horizon (indicating lateral water flow in this layer) directly overlying the clayey subsoil. All soil profiles on hillslopes are acidic to strongly acidic throughout.

While soils on remnant plateau surfaces have similar texture profiles to soils situated on hillslopes (loamy surface soil grading to or overlying clay subsoil), they are nonetheless older and more deeply weathered. Plateau soils contain ironstone gravel, while weathered rock occurs at greater depth compared to hillslope soils. Kaolin clay-rich, deeply weathered material underlies plateau soil profiles; while silty textural grades are less common than on hillslopes and primarily occur in subsoil layers. All plateau profiles are acidic to strongly acidic throughout.

Wetland soils are characterised by texture contrast profiles. They typically have dark-coloured highly organic topsoil—often including a peaty surface of up to 30 cm thick—which overlie low permeability sodic (and dispersive) clayey subsoil with prismatic structure, which in turn grades to soft, highly weathered rock. Organic-rich surface layers vary greatly in thickness, from less than 10 cm to over 1 m. Topsoils (comprising organic rich surface layers and sometimes a bleached subsurface layer) are rarely less than 30 cm thick and have loamy, clay loamy or peaty textures. Clayey subsoils are predominantly grey, blueish-grey or greenish-grey in colour (gley colours of the Color Chart for Gley of the Munsell Soil Color Charts (see Munsell, 1988), indicating anoxic conditions associated with persistent saturation. The presence of peat also indicates conditions where saturation predominates, as organic matter accumulation greatly exceeds decomposition owing to limited oxygen availability (peat is a material that is dominantly composed of organic materials accumulated under wet conditions (see National Committee on Soil and Terrain, 2009).

Subsoil layers exhibiting preferred flow were observed in clayey subsoils and lower soil substrate layers of wetland areas: as preferred pathways within spaces around clayey structural aggregates (evidenced by saturated inter-aggregate spaces containing bleached or iron-stained sandy material); and as layers of saturated clay bounded above and below by layers of moist but unsaturated clay.

Deposition of sediments (probably post land clearance derived) is evident in some wetland areas up to a thickness of around 60 cm. Some organic-rich surface layers also contain charcoal fragments, which seem to derive from early European settlement times when fire frequency and intensity were increased. However, wetland soils are mostly formed in situ from underlying metasediments, with silty texture grades predominating and gley-coloured soft, weathered siltstone soil substrate continuing to considerable depths (e.g. greater than 3 m).

Many wetland areas are obvious footslopes (very lower slopes) that are geologically continuous with adjacent hillslopes. Wetland soils are acidic to strongly acidic in upper layers, with some peats exhibiting extremely low pH (e.g. 4.0), while lower subsoils or upper soil substrate materials are often neutral to alkaline. This is the result of the accumulation of cations (e.g. calcium and sodium ions) from catchment areas (originally derived from dissolved salts in rainfall, sea spray or other precipitation, which ultimately derive from the sea) within wetland soil layers. It was observed in the Foggy Farm wetland that tea-tree (*Leptospermum continentale*) seemed to be associated with peaty soils, an association also believed to be commonly observed in wetlands in the South East of the State. Sodium-rich sodic and dispersive (and hence low permeability) layers in wetland soils derive from the same source. Being closer to the sea, Wither Swamp soils are noticeably more sodic than the soils of the Foggy Farm catchment—this includes both wetland and general catchment soils.

Prior work has found depositional soil profiles to considerably greater depth. For a comparable perched wetland in the adjacent Boat Harbor Creek catchment, Bickford and Gell (2005) identified pollen to depths of 180 cm in a wetland soil core. This depth of soil was evidently built up at that site through peat accumulation and sedimentation over the weathered basement throughout the Holocene period (recognising that these are separate processes, with peat forming in situ, while sedimentation materials are derived from erosion and deposition). Accumulation rates at the study sites appear to have been much lower, or commenced more recently. Alternatively, deeper depositional soils may be present at the sites and were simply not observed owing to local variability. This is unlikely to be the case at Foggy Farm given the intensive soil survey effort. Observed wetland areas generally lacked significant sedimentation as they are predominantly erosional landscapes. Over time erosional materials from catchments appear to have been mostly washed out of catchments, with only limited accumulation in wetlands.

Appendix D presents detailed descriptions of soil landscape map units developed for the Foggy Farm catchment and also includes tabulated soil profile descriptions for soil characterisation sites at each of the three study catchments.

5.2.2. CATCHMENT AND WETLAND SOIL WATER MOVEMENT PROCESSES

Field investigations at the three study catchments indicate that surface and shallow sub-surface processes dominate. Consistent with the Barnett and Rix (2006) conceptual model, no interaction with regional groundwater is believed to occur. Drilling records in the study region indicate that water tables within the Kanmantoo Group sediments are at depths of 60 m or more. This represents a depth to water of perhaps 20 metres below the lowest elevation of wetlands at field sites. Drainage to these deep regional groundwater flow systems is thought to be extremely limited and this was considered to be a negligible component of the catchment water balance and not represented in modelling.

Perched wetland hydrology can in general be characterised as a dynamic equilibrium between the seasonal accumulation and concentration of catchment rainfall excess via overland flow and saturated soil water movement and its loss via streamflow, horizontal seepage and evapotranspiration during the months where a rainfall deficit exists. Although not individually quantified, it is likely that overland flow would comprise a large proportion of the total runoff from all catchments. However this is limited in duration to rainfall events and in order to understand the persistent saturation that creates wetland conditions it is necessary to understand saturated soil water movement processes.

Three pathways for saturated soil water movement to the wetland have been observed and their relative importance seems to largely determine wetland hydrology, at least in terms of the duration of saturation. Two of these pathways have previously been quantified in a range of investigations in the Mount Lofty Ranges (Smettem *et al.* 1991; Leaney *et al.* 1993; Stevens *et al.* 1999; Fleming and Cox 2001; Cox *et al.* 2002; Bestland *et al.* 2009) and, in addition to overland flow, are thought to be processes common to all perched wetland catchments. The third saturated soil water movement pathway was not known prior to the study and is of unknown prevalence in perched wetlands in the region. With reference to Figure 23, the three identified saturated soil water movement processes are discussed below.

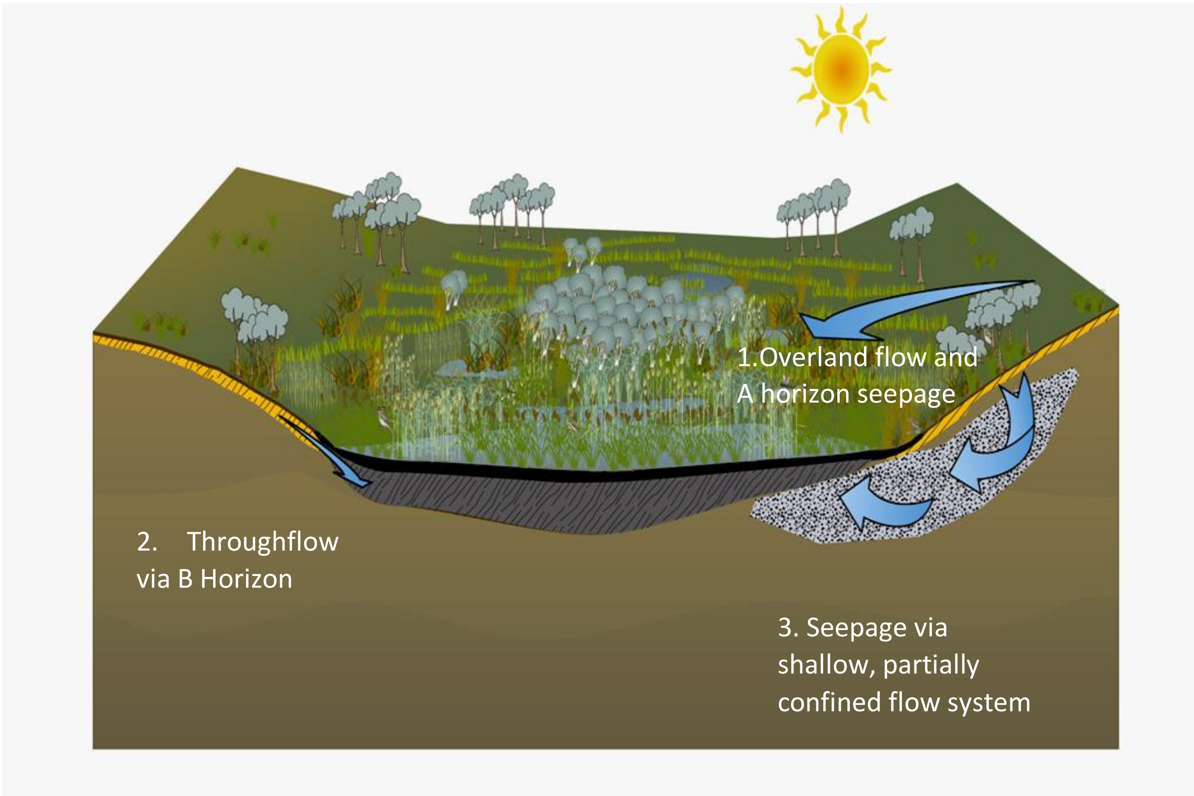


Figure 23. Conceptual model for a perennial perched wetland

1. Water movement to wetlands via throughflow in A horizons

Pathway 1 in Figure 23 represents horizontal movement through A horizons over the clayey subsoil. In all wetlands, there was significant evidence of A horizon throughflow: (i) the obvious porosity of organic rich A horizons being one main body of evidence, while (ii) bleached subsurface layers provide the other main evidence. Evidence for A horizon throughflow in hillslope soils was only seen on lower slopes where bleached subsurface layers were evident, but this does not discount A horizon throughflow in other hillslope soils. It is possible that bleached layers were not evident as fluxes are lower, or because there has been insufficient time since formation for bleached layers to develop.

Depths to control (high clay content) soil layers were typically less than 40 cm in general catchment soils, often much shallower, although wetlands soils were deeper (Appendixes D, F). As with wetlands, vertical drainage is impeded in general catchment soils because of this high clay content, the relatively poor structure of subsoils and the closely-packed nature of the underlying weathered rock. Relatively poor subsoil structure is characteristic of these areas and is largely a result of clay mineralogy, where kaolinitic clay minerals are dominant (Hall *et al.* 2009).

Vertical movement of water is also restricted (especially in wetlands) as clay B horizons are sodic and dispersive, hence of low permeability, encouraging A horizon throughflow. Some general catchment subsoils are affected by sodicity. Wither Swamp is particularly affected—the closer proximity of this catchment to the sea (and hence to prevailing winds, sea-spray and rain bearing higher levels of salts) the probable cause. The influence of the shallow sodic dispersive layer on vertical infiltration of moisture can be clearly seen in the continuous soil moisture data from the Wither Swamp (Appendix I). As detailed soil survey work was not conducted at Springs Road, the extent of sodicity in general catchment soils is largely unknown, however, all of the three soil characterisation sites outside of the wetland had sodic and dispersive soil layers.

Owing to the shallow nature of the general catchment soils and relatively high hydraulic conductivities (Appendix F), this pathway is very much coupled to seasonal climatic conditions. Waterlogging of surface soils adjacent to wetlands to a maximum of about 20 cm depth was directly observed only during late winter and spring. As water moved via this pathway only during the wettest months, this does not offer an explanation for the persistent saturation of wetland sediments observed into summer and autumn.

2. Water movement to wetlands via throughflow in B horizons

Water movement via macropores and cracks within B horizons is identified in Figure 23 as pathway 2. Water infiltrating to the clay-rich lower soil layers moves via preferred pathways such as spaces between soil aggregates and other preferred pathways such as macropores created by old root channels. The presence of fresh bedrock at depth constrains vertical infiltration resulting in more or less horizontal water movement according to elevation gradients.

This process was observed at the native vegetation catchment in spring 2009 adjacent to the wetland at a time that wetland sediments were fully saturated and shallow surface water was present. Following the observation that a lower slope soil-access tube had unexpectedly filled with water, hand augering was undertaken into the weathered basement about 15 metres from the wetland edge and 1.5 m elevation above the surface water level of the wetland. At around 80 cm depth water was observed to freely drain into the hole from cracks in the subsoil. Hence at the Springs Road site it was apparent that preferential flow paths in catchment subsoils were contiguous with those of the wetland, providing a direct hydraulic pathway to wetlands. As this pathway was not identified until late in the project and installed monitoring was not designed to measure soil movement fluxes through the different horizons, no estimates of the volumetric importance of this contribution to the wetland water balance at the site was possible.

No subsoil saturation was observed outside of wetlands at Foggy Farm at any time. At Wither Swamp, saturated water intersected on slopes adjacent to the wetland was under pressure and thought to be the product of a different pathway (referred to as pathway 3 and discussed below). Hence this process was only directly observed at one site and the prevalence in perched wetlands more broadly is unknown.

B horizon throughflow does offer a possible explanation for permanent springs once observed as common in the region prior to clearance of native vegetation. In a Department of Mines publication on the geology of the region during the 1950's, the presence and character of numerous springs in the Parawa area is described (Campana and Wilson 1953, pp. 22–3):

> *In the portions of the elevated plateau areas now in process of being cleared ..., numerous springs have been recognised.... They form soaks at the head of the creeks which drain and dissect the plateau slope ... The water-bearing stratum is an argillaceous (clay-rich) deeply weathered and frequently lateritized (containing ironstone gravel, which is sometimes cemented) material formed in situ at the expense of the underlying Kanmantoo formations, the soaks occurring at or near the contact of the unweathered country-rocks. ... flow is ordinarily small (of the order of 500–5000 gallons daily, at most, during the summer season), but it is generally permanent depending on the extent and retentive power of the water-bearing blanket. The good drainage conditions allow the free movement of water in the store and are responsible for a low degree of salinity. ... The flow may be increased by adequate traps, such as underground drainage ... or open channels in the swamps which form downstream ...*

Based on field observations it would seem likely that these springs were discharges from B horizon throughflow pathways as observed at Springs Road. It is of interest that this description refers to the occurrence of numerous permanent springs and the presence of swamps forming downstream of these in a largely pre-cleared landscape. While wetlands are well mapped in the area and clearly seasonal springs are common, the prevalence of permanently flowing springs in the contemporary landscape is not known for certain but has not been observed to be common.

At the discharge rates reported (flow rates of 500–5000 gallons per day over summer) the Campana and Wilson (1953) springs would not maintain permanent surface saturation across an entire wetland as observed at the Wither Swamp site. For example, considering the wetland at Foggy Farm at around six hectares area, even 5000 gallons per day (~23 kL/d) would only equate to around 0.02 mm/d if distributed over the wetland surface. This is well below the evapotranspiration demand over summer. Many, if not most, perched swamps are considered by staff of the Fleurieu Peninsula Swamp Recovery Program to have only 'core areas' that remain permanently saturated (A. Stevens (FPSRT) 2008, pers. comm., 7 March). Flow rates of this volume would support saturated areas, suggesting headwater gullies may be critical areas for perched wetland hydrology and that pathway 2 may be a water movement and storage process capable of supporting such areas of persistent saturation in perched wetlands.

B horizon throughflow is a well researched water movement pathway in Mount Lofty Ranges texture contrast soils (Smettem *et al.*, 1991; Leaney *et al.*, 1993; Fleming and Cox 2001; Cox *et al.* 2002; Bestland *et al.* 2009). By separately quantifying the rainfall volumes transmitted via throughflow in the A and B horizons and overland flow, these studies have found the relative importance of each changes considerably. Differences are described from one site to another site (e.g. compare Stevens *et al.* 1999 and Fleming and Cox 2001); for different A and B horizons (Bestland *et al.* 2009); for different soil types, topography and positions on the slope profile (Cox *et al.* 2002); under different rainfall regimes (Cox *et al.* 2002); and between seasons within the same site (Chittleborough *et al.* 1992). Studies cited have typically focussed on nutrient transport pathways and have not considered the influence of throughflow on the hydrology of a receiving water body. Bestland *et al.* (2009) determined the contribution of B horizon throughflow to ephemeral streamflow and found contrasting results within a catchment that were attributed to subsoil texture. Clayey duplex soils (comparable to catchments in this study) underlying an area of the catchment cleared for grazing yielded rapid and shallow throughflow; while sandy subsoils under native vegetation had a larger vadose-zone storage and introduced a lag to the runoff response.

The findings of Bestland *et al.* (2009) suggest water movement via B horizons would not contribute to prolonged saturation in wetlands, but a study of these processes in clayey duplex soils under remnant native vegetation is not known to have occurred. The effects of the remnant deep rooted perennial native vegetation at the Springs Road site may favour infiltration and increase the retention time in B horizon pathways. Changes to subsoil secondary porosity since clearance may have altered fluxes via the different pathways. In particular, any contour ripping that disturbed subsoil structure would be expected to profoundly affect B horizon throughflow, likely reducing the contribution of this pathway to runoff and increasing overland flow.

Evidence suggests that wetlands supplied only through overland flow and soil water movement pathways 1 and 2 would still be expected to be only seasonally saturated across much of their surface area based on this study. However, an additional pathway was identified at the Wither Swamp which explained the permanent saturation at the wetland surface and perennial streamflow observed at the site.

3. Water movement to wetlands via locally confined flow systems

Throughflow in the A and B horizons (pathway 1 and 2 in Fig. 23) described above are consistent with the original Barnett and Rix (2006) model. Although only shown entering from the side of the wetland in Figure 23, water would potentially move via both pathways from all upstream directions. The contribution from convergent, concave slopes such as headwater gullies are likely be the dominant source of water for both of these processes and for overland flow. Perennial perched wetlands such as the Wither Swamp are likely influenced by the same processes, but an additional water balance component was also evident at this site (pathway 3 in Fig. 23).

This third saturated soil water movement pathway was observed only at the pasture study catchment—The Wither Swamp, which is known to have yielded perennial stream flow since at least the 1960s (P. Filsell 2007, pers. comm.). Initially hypothesised to be a result of the pasture land use, field investigations demonstrated that the site has an unusually high yield (runoff coefficient of 0.31 for 2009). The permanent surface water flow and saturation at the wetland surface (Figs 21 & 22) is evidently supplied from a confined local groundwater flow system underlying slopes adjacent to the wetland. The aquifer medium in this case is fractured quartz rock and fine sand, but any unconsolidated material within the weathered basement matrix would potentially function in an analogous manner. This (presumed to be lens or similarly shaped) formation is embedded within the weathered basement, such that it becomes nearer to the ground surface at lower points on the slope, ultimately discharging near the break of slope. The deepest known point for the upper surface of the water-bearing layer is around three metres vertically below mid slope elevations (5–10 m elevation above the wetland).

Discharge from the flow system was observed to only the northern side of the wetland at an elevation of about 1–2 m above the wetland surface (via argillaceous 'spew holes'). It may also seep upward directly into wetland sediments under the small pressure head, either via macropores or where the overlying weathered basement is only partially confining. An observed transverse salinity gradient in surface waters across the wetland opposite the seepage zone suggests that lateral inflows from only one side of the wetland dominate. The quartz formation was not observed to outcrop at the surface and was only detected through hand augering within the seepage zone at footslopes. It is theorised to recharge via preferential pathways (e.g. macropores) at hilltops and become confined as it flows down gradient beneath the clay-rich, often sodic subsoil. No direct evidence for this was observed however.

This additional pathway for water movement is not subject to evaporative losses until discharging to the wetland surface. This introduces a lag period in which wetland hydrology is decoupled from the climate, where stored winter excess rainfall is gradually released to the surface. This supports the observed year-round saturated conditions at the wetland surface (Fig. 22 hydrograph) and perennial streamflow.

The resulting variations in surface water yield are considerable (Sect. 5.1)—the Wither Swamp surface water site downstream of the wetland gauged 375 ML of runoff in 2009. This is equivalent to around 250 mm depth of runoff, an increase of over 50% compared with an estimated yield under pasture based on the runoff depth observed upstream of the wetland extrapolated across the entire catchment area (162 mm – Sect. 5.3). Groundwater hydrographs from the Wither Swamp maintained a virtually constant level at the ground surface, deviating less than 5 cm over the year (Fig. 22). Contrasting this, Springs Road groundwater levels varied more than 1 m over the year, while the Foggy Farm wetland dried completely, a depth to water variation exceeding 2 m (Figs 22 & 27). Hydrographs from any perched wetland fed by an analogous confined local-flow system should be readily identified via reduced coupling of groundwater levels to seasonal climatic variation.

As this flow system was unknown prior to this study, quantifying fluxes were outside experimental design. As a result any changes to this component of the water balance resulting from different land use has not been investigated. Logically deep rooted perennial plantings adjacent to discharge zones will directly access subsurface seepage below the root zone of pasture, inevitably affecting wetland hydrology. Deep rooted perennials would also reduce recharge through increased interception and direct root zone moisture usage—this may have an even greater impact than the additional transpiration losses at discharge zones. Whether pathway 3 type flow systems would support permanent saturation in wetlands with catchments under a deep rooted perennial land cover is probably unlikely, but remains unknown. Widespread clearance of native stringybark woodland occurred in the region largely during the 1950s (Bickford *et al.* 2008). This would have increased recharge to any formation analogous to that found at the Wither Swamp, increasing runoff volume and in particular duration (and wetland hydro-period) over and above any increase expected in changing land use from woodland to grassland.

Conversely, changes to fluxes from such systems in a situation of afforestation of former pasture are unknown, but may have already occurred. Local landholders recall that prior to the pine plantation being established in 1989–90, Foggy Farm wetland had perennial stream flow (K. Bartolo 2009, pers. comm.). The possibility remains that the springs reported from clay-rich soils in Campana and Wilson (1953) and theorised herein to be due to throughflow via B horizon cracks and macropores could produce perennial stream flow. This would not be predicted using the modelling approach adopted here, but subsoil processes were not modelled explicitly and this remains a possibility. In particular, any changes in the continuity of cracks and macropores resulting from ripping the subsoil in preparation for planting also remain unknown, but would clearly have a profound effect on subsoil water movement. Interruption of process 3 recharge or discharge at the site as a result of afforestation would also offer an explanation, but while considerable soil survey work (including profile descriptions and mapping) was undertaken at the Foggy Farm catchment, no evidence was found of any quartz-sand lens analogous to that observed at Wither Swamp.

5.2.3. WATER MOVEMENT AND WATERTABLE DYNAMICS IN WETLANDS

Within wetland sediments water also moves via analogous pathways to those presented in Figure 23 and discussed above—saturated flow occurs through cracks and macropores in subsoils (Appendix D) and via matrix flow through topsoil layers with excess water moving through the system as streamflow. Where topographic or other impediment of down-valley water movement via throughflow B is impeded, formation of perched wetlands appears to be favoured. The extent and duration of surface saturation in the wetland will be influenced by a number of factors of which land use is only one.

In wetlands however water-movement pathways become more complicated and subsoil and lower layers that were saturated (or almost so) were at times observed to be sandwiched between other clayey layers that were not wet. This suggests saturated flow paths through layers of lower resistance are important in the movement of water through wetlands. This three dimensional layering creates a complex hydrology that was not able to be adequately quantified by the simple monitoring networks or modelling approaches adopted for this study. Future work will need to develop some means to quantify fluxes and relative storages within these layers and to model these responses to improve estimates of wetland hydro-ecology.

Preferential flow paths through cracks and macropores in wetland subsoils can be likened conceptually to water movement in a fractured rock aquifer, with a relatively impermeable matrix. Adopting fractured rock aquifer terminology, this is referred to as secondary porosity. The storage volume associated with the cracks and macropores comprising secondary porosity was estimated at the two seasonal wetlands by observing the change in standing water level to the depth of early season rainfall events. Figure 24 shows the rapid rise of groundwater levels in response to two such events, indicating rapid direct infiltration of surface soils and a rise in wetland water level. Catchment soils had not yet reached saturation and no runoff is believed to have contributed to the rise in water level, while any throughflow would be negligible based on the ambient rate of rise prior to the events. Hence by comparing the increase in water level within the wetland during the event to the measured depth of rainfall for each, estimates of the storage volume (secondary porosity) were obtained.

For two early season events at three wells across the two seasonal wetlands an average value of 0.12 ± 0.02 (standard deviation) for secondary porosity of subsoils was obtained. Assuming uniform distribution and 1 m of depth where saturated conditions dominate subsoil, this represents around 1.2 ML of storage for each hectare of wetland area in the top metre of wetland subsoils. Gleyed soil profiles were observed to 3 m depth (maximum achievable with hand augers) and while evidence of saturated conditions was often observed at depths greater than one metre, no data were available to estimate the secondary porosity at these greater depths. It is probably reasonable to assume that both the volume and connectivity of the secondary porosity will decrease with depth.

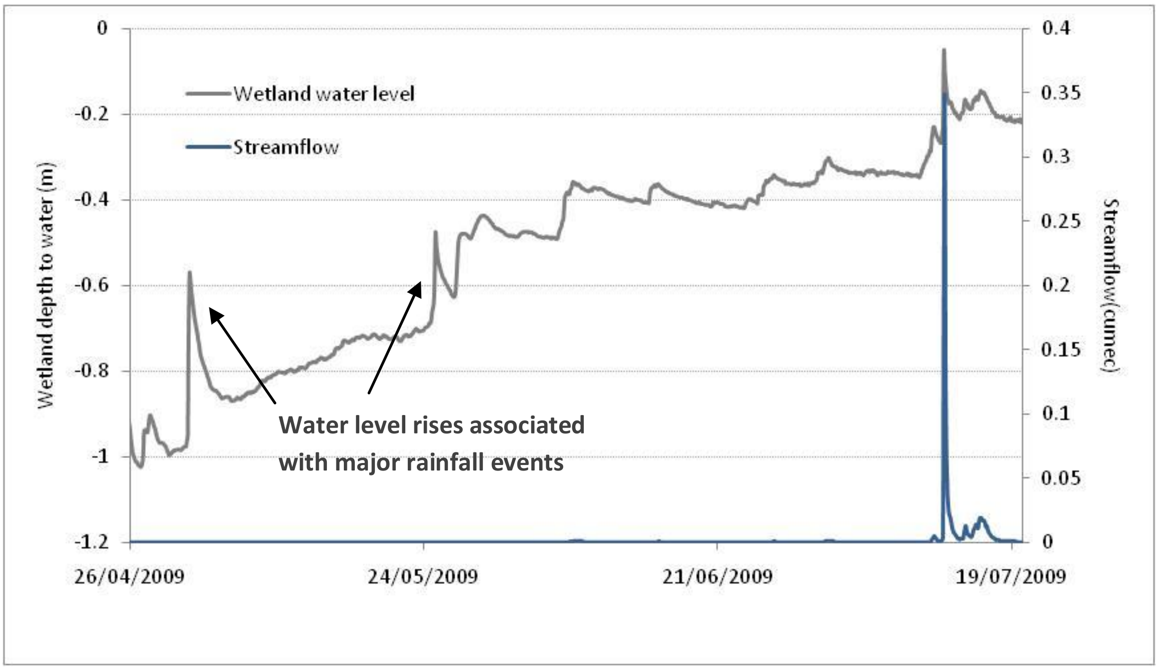


Figure 24. Hourly hydrographs for downstream piezometer and streamflow, Springs Road wetland, 2009

Field data matching observed groundwater and stream flow hydrographs shows that wetland subsoil stores need to fill prior to overland flow being generated from the catchment (Figs 25 & 26). Once breaking rains occur, water rapidly enters B horizon pathways as shown by the hourly groundwater hydrograph in wetlands (Fig. 24). Overland flow does not occur until wetland B horizon stores are filled and A horizons are near filled (Figs 24–6). At the seasonal wetlands this requires a depth of accumulated rainfall of around 200 mm. Chittleborough *et al.* (1992) observed similar rapid infiltration to B horizons in catchment slopes, while Bestland *et al.* (2009) observed a 200 mm depth of autumn rainfall accumulated in catchment soils prior to streamflow commencing in 2007 at a comparable ephemeral catchment with texture contrast soils. Neither of these studies incorporated processes in wetland areas however.

In terms of the streamflow leaving the catchment, this initial storage demand is analogous to the effects of an on-stream dam. This is how wetlands were represented in modelling, producing a greatly improved calibration to observed data over that achieved without this store. The size of the wetland storage is a key factor in adopting this approach and for this study it was derived based on a combination of measured soil hydraulic properties, observed depth of saturation and mapped surface area.

Seasonal breaking rains and catchment inflows initially fill the wetland storage to the point where any additional water arriving at the wetland will leave as streamflow, which includes a component of throughflow from wetland surface layers. As discussed above, wetlands are believed to occur at places in the landscape where throughflow via the B horizon is constrained (or even prevented from draining) via topography or an unspecified sub-surface water movement barrier (for example an impermeable rock formation). Water moving out of the wetland via B horizon throughflow was limited to watercourses and was only observed during periods of streamflow and this loss pathway is thought to be limited in terms of total volume at the seasonal field sites. Hence the main runoff generating process for perched wetlands appears to be saturation-excess overland flow when wetland (and catchment) subsoil stores are filled, with flow persisting until wetland A horizons have largely drained.

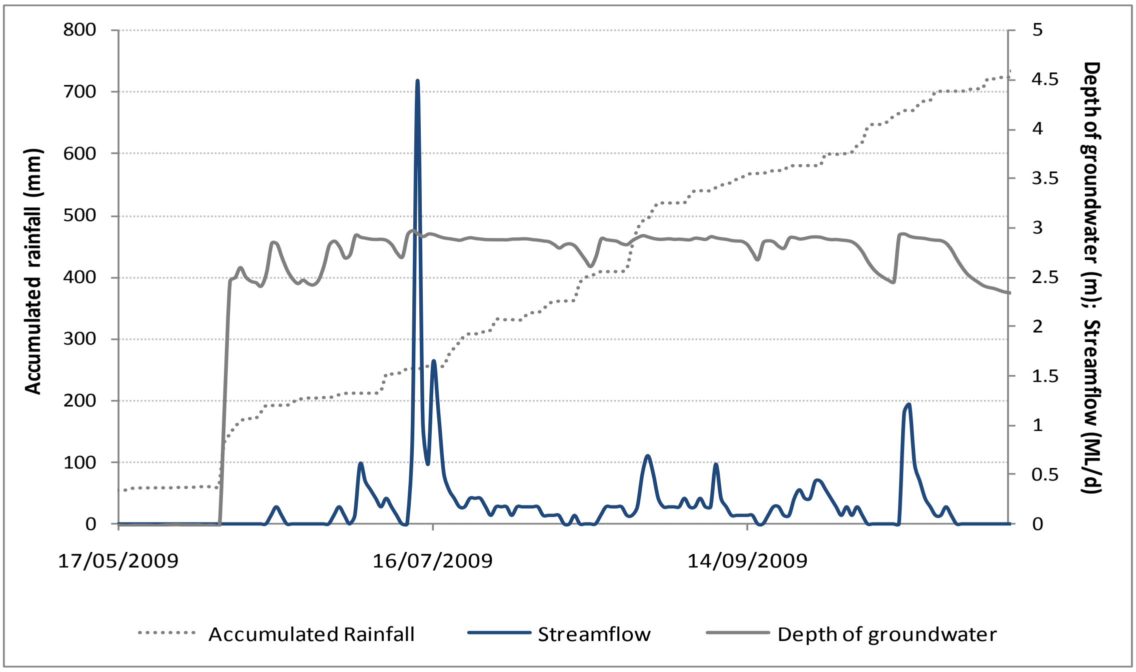


Figure 25. Daily accumulated rainfall, wetland watertable and streamflow at Foggy Farm – May–Oct 2009. Note depth of groundwater is the depth above the logger. Ground level is at 2.9 m depth.

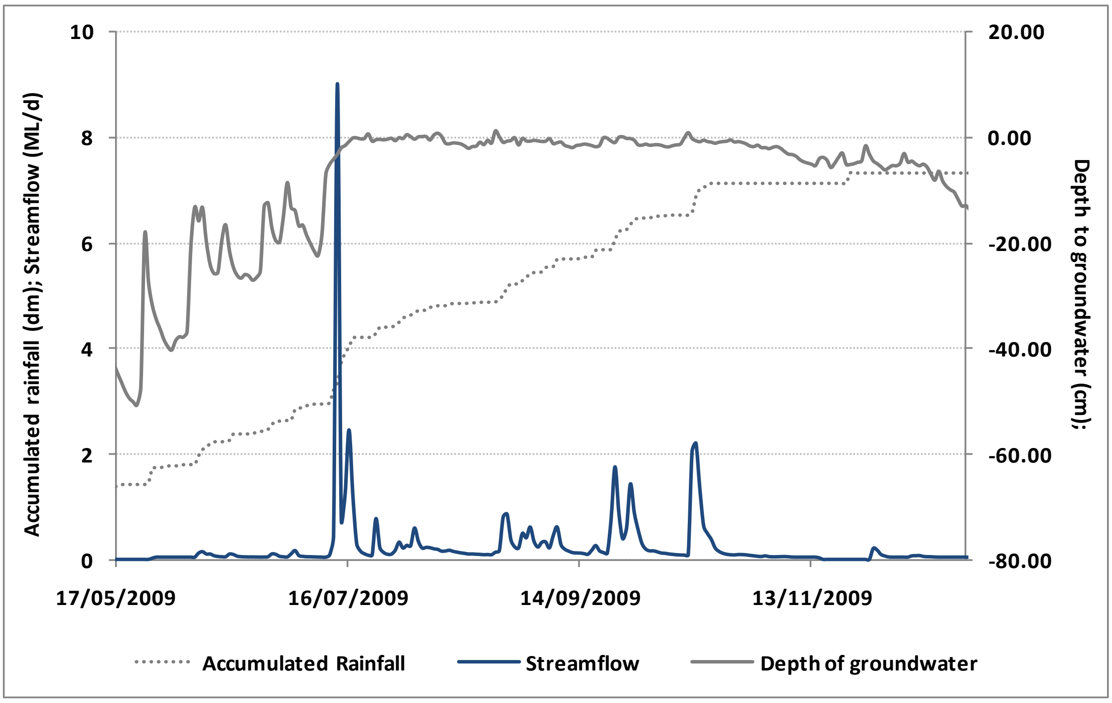


Figure 26. **Daily accumulated rainfall, wetland watertable at the upstream piezometer and streamflow at Springs Road 2009**

Wetland soil stores remain saturated only as long as water moves through pathways 1 (which includes overland flow and flow through surface litter layers), 2 or 3 to the wetland (Fig. 23). By late spring and through summer to autumn months at seasonal wetlands, streamflow ceased as A horizons drained completely, but water still remained in wetland residual stores (Figs 25–27). Once rainfall no longer exceeds evapotranspiration, residual storage then declines at a rate dependent upon whether inflow to the wetland continues to occur via B horizons. For Foggy Farm, no replenishment of daily evapotranspiration losses was observed in the hydrograph and the wetland residual store dried in a rapid and broadly linear fashion (Fig. 27). The Springs Road wetland hydrograph showed clear patterns of daily decline and overnight rise in watertable level, indicating water was still moving to the wetland from surrounding hill slopes (Fig. 28).

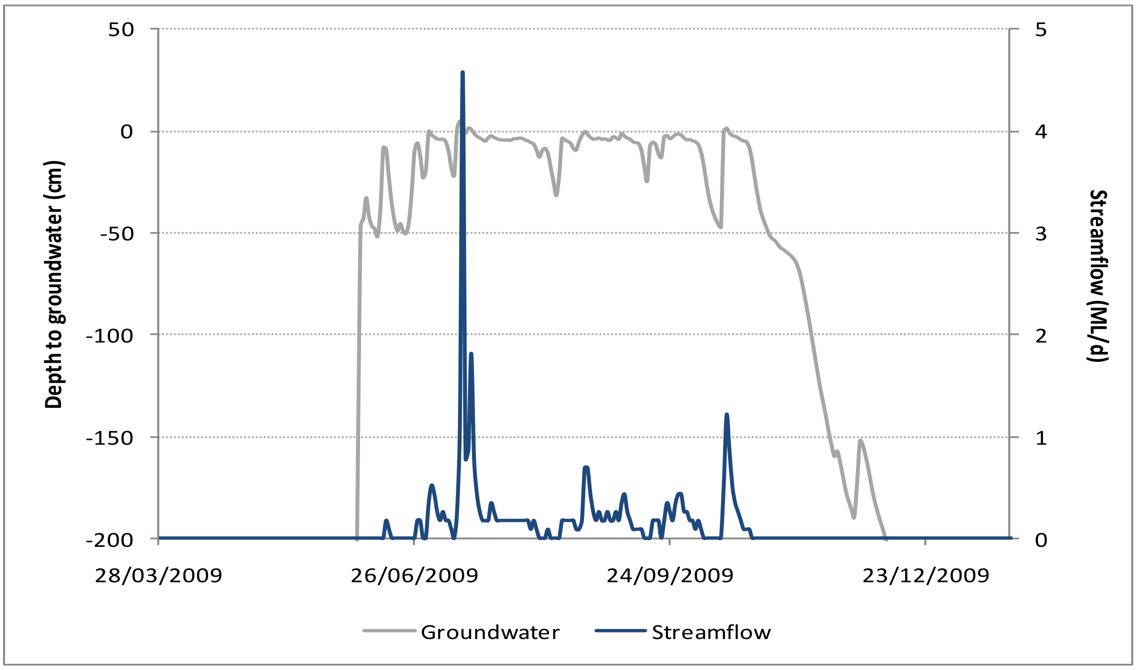


Figure 27. Daily streamflow and depth to groundwater, Foggy Farm wetland 2009

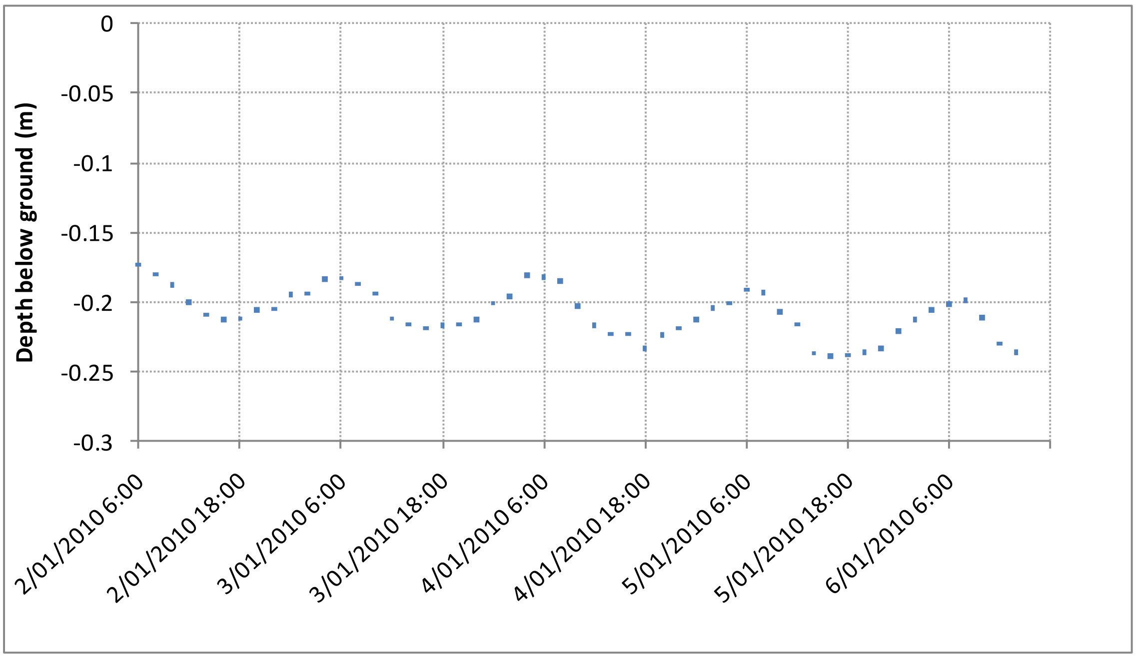


Figure 28. Diurnal watertable fluctuation - Springs Road wetland downstream piezometer

Interpretation of field site groundwater hydrographs

The dynamics of the watertable depth-duration relationship at each perched wetland can be expected to vary dependent upon:

- the presence and relative importance of the different contributing soil-water movement pathways 1–3 described above, both within wetlands and any extent into the surrounding catchment
- the different ratios of wetland storage volume to contributing catchment area
- catchment land use and land management.

Based on the theorised pathways, field observations and hydrograph responses, it is possible to hypothesise the relative importance of the described pathways at the study catchments. The Foggy Farm wetland is believed to currently rely largely on direct rainfall and throughflow in the A horizon, with either limited B horizon water movement, or rapid transmission of water through this layer, resulting in no lag being introduced to hydrology. No other confined pathway for water movement such as that observed at Wither Swamp is believed to be present. Evidence for the lack of B horizon throughflow-contribution from the catchment is the lack of persistence in the saturation of the wetland. The drying trajectory of wetland hydrographs indicate that water removed during the day by evapotranspiration was not being replenished during the evening as would be expected if B horizons drained more gradually as appears to occur at the Springs Road site.

Springs Road catchment is theorised to represent both A and B horizon throughflow, with the latter representing the pathway supporting persistent saturation and continuing supply of water to the wetland over summer (Fig. 28). Infiltration to subsoils may be facilitated by the presence of deep-rooted native vegetation and there is also a greater surface plateau area for this catchment, which may be important. Watertables in the wetland remained persistently at the surface over the winter spring period, before declining as evapotranspiration demand exceeded supply (Figs 26 & 28). The wetland was not observed to dry completely however.

The Wither Swamp hydrograph does not deviate from the wetland surface (Fig. 22), illustrating the climatic decoupling believed to be entirely a result of the additional subsurface pathway for water delivery to the wetland (pathway 3, Fig. 23). It is of interest that only pathways 1 and 3 were observed at this site and no direct evidence of B horizon throughflow was observed outside of wetland sediments. Soil moisture data and surveys indicate that dispersive clays at the top of the B horizon greatly limit vertical infiltration. Overland flow is likely to produce a large proportion of total runoff as A horizons are shallow and will saturate rapidly (and shallow surface soil waterholding capacities are generally limited). Saturation persisted on lower slopes for almost six months, indicative of the perched saturation within A horizons (for example, at soil characterisation site CH176 which exhibits a bleached A2 [or subsurface] horizon directly overlying a sodic clay subsoil starting at 33 cm [see Appendixes D & I]). The poor drainage from shallow soils raises the question of how recharge occurs to the confined local flow system (pathway 3, Fig. 23). It is presumed that macropore drainage at hilltops bypasses the soil matrix, infiltrating to the quartz-sand lens where it then moves horizontally becoming confined.

While not explicitly modelled, the effects of different land uses on pathway 1 and 2 are incorporated within this study. Modelling methods were unable to distinguish between these two processes and represent an integrated simulation of land use impacts on both pathways.

5.2.4. PLANT AVAILABLE SOIL WATER AND ROOT DEVELOPMENT

Wetland vegetation

For much of the year, over much of the surface of the two seasonal wetlands, soils were free draining in a lateral sense and therefore seasonally well oxygenated, at least under climatic conditions prevailing during this study (Figs 26 & 27). Plants, including invasive species without the convective flow mechanisms to oxygenate roots more typical of wetland vegetation (e.g. Brix *et al.* 1992), are not precluded from dispersing to wetland sediments. The proximity of agricultural lands means seed stocks of ruderal and r-selected annual agricultural weeds are highly prevalent. Such plants can potentially establish under optimal conditions as wetland watertables decline in spring and complete a growth and flowering cycle prior to the following seasonal period of inundation. Decreases in the duration of seasonal saturation and disturbance of wetland vegetation would be predicted to create conditions that would increase the proportional importance of this vegetation component within wetlands.

In addition to drainage characteristics, the structure and chemistry of catchment and wetland soils has consequences for plant root development, in turn influencing wetland hydro-ecology. Schenk and Jackson (2002), in a global synthesis of root distribution data, state that for > 90% of species globally at least 50% of roots are found within the top 0.3 m of the soil profile. Soil moisture variations (Appendix J) suggest wetland soil root distributions at field sites have a much higher proportion of roots in surface soils. At soil characterisation and description sites, root density was estimated for each soil horizon (according to National Committee on Soil and Terrain 2009). In general, the wetland sites showed abundant surface soil root growth to greater depth (to around 30 cm) than is the case in general catchment soils, owing to thicker organic-rich and friable surface layers. Also, more roots than is normally the case in non-wetland soils were observed in subsequent layers. In addition, easily detectable roots continued to much greater depths than at general catchment sites, generally to at least 1 m and in some cases to 3 m.

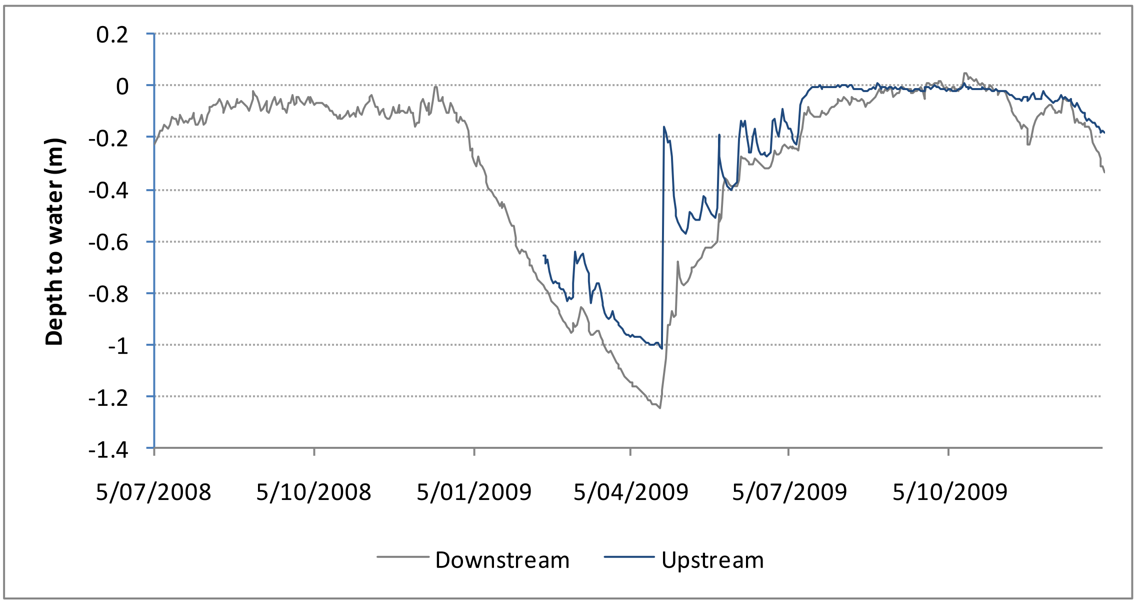


Figure 29. Daily groundwater hydrographs for Springs Road wetland piezometers July 2008–Dec 2009

Although some roots were evident at depth in wetlands, soil strengths likely constrained root development both in the wetland and certainly within the surrounding catchment. Penetrometer strengths for wetland soils at field sites measured at 0.3 m depth averaged almost 6 MPa, despite many soil samples being near to or above field capacity (see Tables 2 & 3 in Appendix F). Soil strengths in this range would be considered to inhibit root growth except within cracks (Appendix F). In hard soil layers, soil strength is more problematic during drier conditions, while aeration is a problem when near field capacity. Root densities, even for plants with a woody habit, were qualitatively observed to be largely restricted to within the surface layers of wetland sediment and roots observed in both catchment and wetland subsoils were often observed to follow preferential layers or cracks, in places growing quite flattened root structures.

Although hydric soils in wetlands were of the order of 2–3 m deep, subsoil secondary porosity was estimated at only around 10% of soil volume. Hence subsoils do not represent a large storage for root water access even if roots are able to penetrate via preferential pathways. In contrast, the porosity of peat and organic-rich loam surficial layers in wetlands varied between 0.65 and 0.75 (Appendix F). At all sites organic-rich layers in wetlands varied greatly in thickness—this layer holds a large proportion of plant available water and clearly comprises an important soil moisture store (Appendix J). The development and maintenance of saturated conditions within surface layers is evidently important in structuring perched wetland vegetation (Sects 4.2 & 5.4). At field sites for the current study, organic-rich layers within wetlands typically did not exceed 60 cm thickness (Appendix D), yet despite this contain the largest soil moisture reserves for plant roots. Topsoils in many cases also included bleached subsurface layers indicating high levels of horizontal water movement consistent with the observation that streamflow coincides with near surface saturation and indicating the importance of shallow water dynamics for wetland hydro-ecology. Isolated cases of greater organic soil depths exceeding 1 m were also observed and this storage clearly varies considerably within and between wetlands. Bickford (2005) for example found depositional soils indicated by fossil pollen in a perched wetland close to Foggy Farm to depths of 180 cm and this layer would presumably be of similar hydraulic characteristics to study wetlands.

Catchment vegetation

Root depths in catchment soils are limited by the relatively shallow depth to underlying weathering rock, while the sodic clay subsoils tend to limit root abundance. The extractable soil moisture from the top metre on the catchment slopes at Foggy Farm for 2009 was estimated from monitoring data to be around 110 mm (Appendix J). A comparative estimate was also made using the available water by soil texture table given in Maschmedt (2002) and using a rooting depth of 90 cm[3]. This approach yielded the same value for the upper slope soil characterisation site (CH179) at Foggy Farm, suggesting this is a robust estimate.

Soil volumetric moisture data at the site also indicated that no substantial change in soil moisture occurred during the period between February and April 2009 (Appendix I). Although some moisture was clearly available or widespread dieback would be expected, this indicates that by late summer, easily accessible soil moisture stores were exhausted and pine trees were likely restricted to active growth opportunistically following rainfall events. It may be unlikely that transpiration (and therefore growth) is soil-water limited every year, given the relatively high soil water store (Maschmedt 2002; Hall *et al.* 2009), but clearly soil moisture stores do limit evapotranspiration (and therefore growth) despite high plant available water and annual rainfall.

3 This is based on an estimated depth for pines, but would be lower for annual pasture plants

5.3. MODELLED PERCHED WETLAND HYDROLOGY

5.3.1. ANNUAL VARIATION IN MODELLED SCENARIOS

Each scenario was run for a simulation period of 37 years (1973–2009), intended to replicate one cycle of a pine plantation growth phase from planting to harvest and incorporating the early cycle higher runoff period. This section presents findings only to provide an indication of variations in annual response, while results for annual stream flow under all scenarios are presented in Appendix A. In addition a generalised rainfall–runoff response for selected scenarios is presented in Appendix B.

Median modelled annual runoff under pasture for the 37-year scenarios was 162 ML and as the hypothetical catchment was 100 ha—this also represents the depth of runoff in mm. Mean annual runoff for the same period was 164 ± 84.5 ML (standard deviation) and the mean annual rainfall for the modelling period was 947 mm. The 100% pine (20 m buffer) and native vegetation (stringybark closed woodland) land use scenarios produced the least runoff, with mean and median statistics being of the order of 25–35% of the pasture land use response (Fig. 30). Mean modelled runoff under pine at full catchment planting with 20 m buffer is higher than native vegetation, although the latter produced a higher median.

All other land use combinations, planting fractions and buffer widths produced less runoff than pasture. Runoff reduction was found to be closely correlated with the percentage area planted (Figs 31 & 32). For both median and mean statistics the relationship between percentage planting and predicted runoff is strongly inverse linear, with slopes of -1.22 and -1.03 respectively (Figs 31 & 32). Each 1% increase in planting fraction was estimated to reduce median runoff over a full rotation by 1.22 mm (0.8%) and mean runoff by 1.03 mm (0.6%) (Fig. 31).

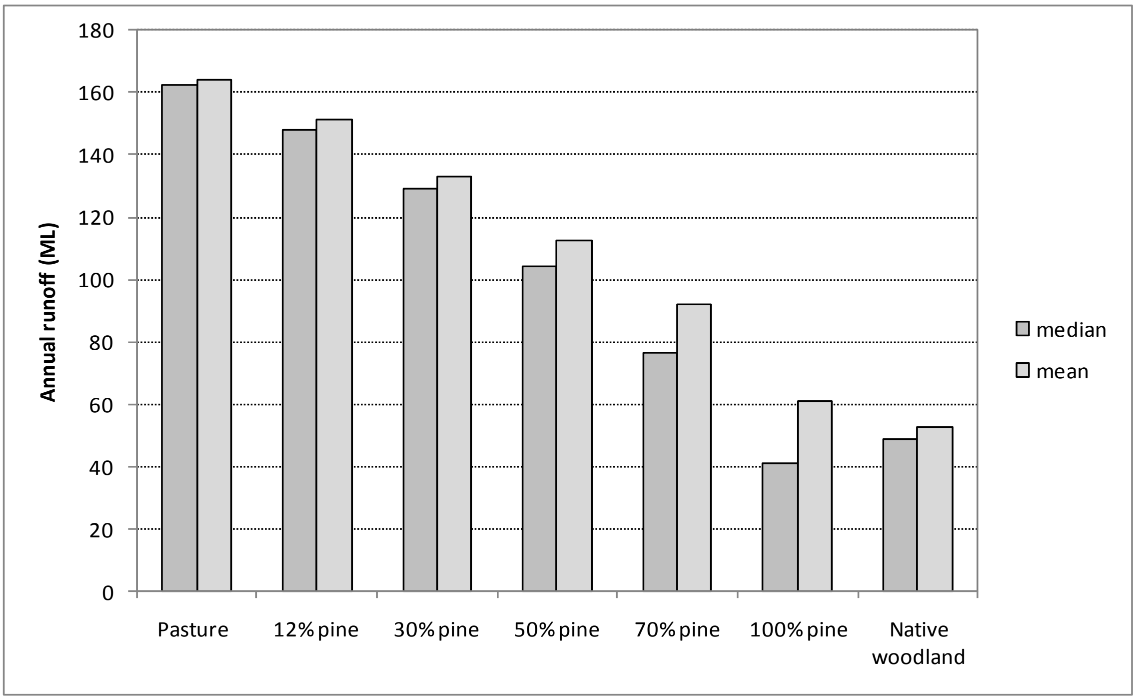


Figure 30. Modelled median and mean annual runoff over a 37-year period for various land uses (all pine scenarios employ a 20 m buffer)

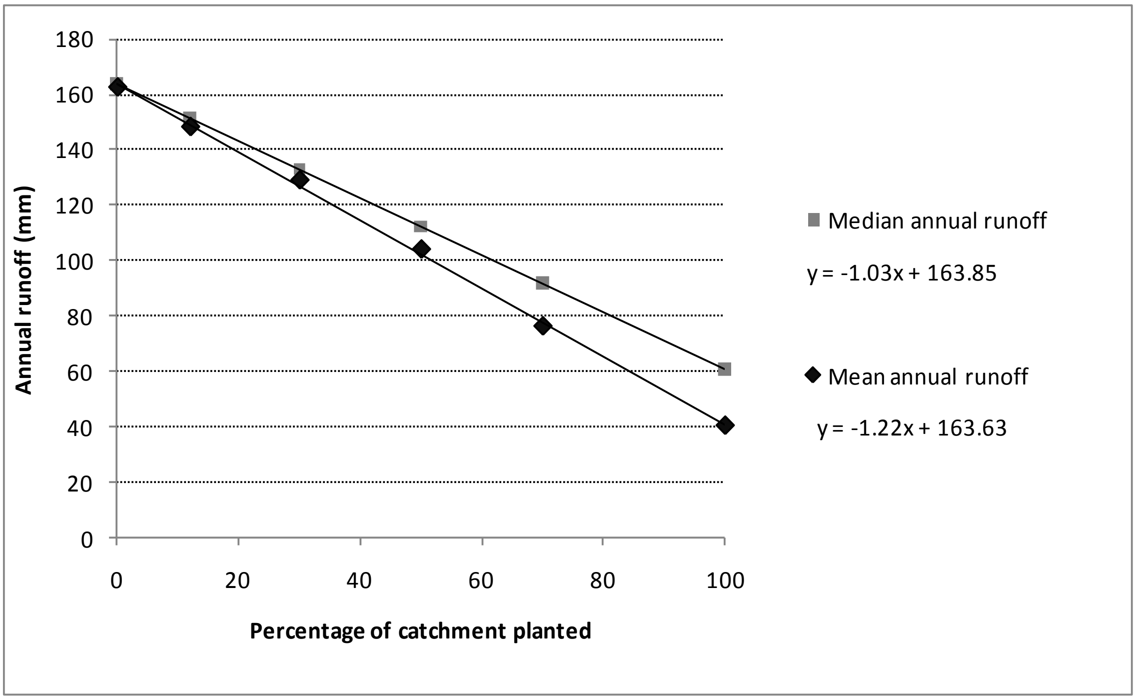


Figure 31. **Modelled mean and median annual runoff as a function of percentage catchment planted**

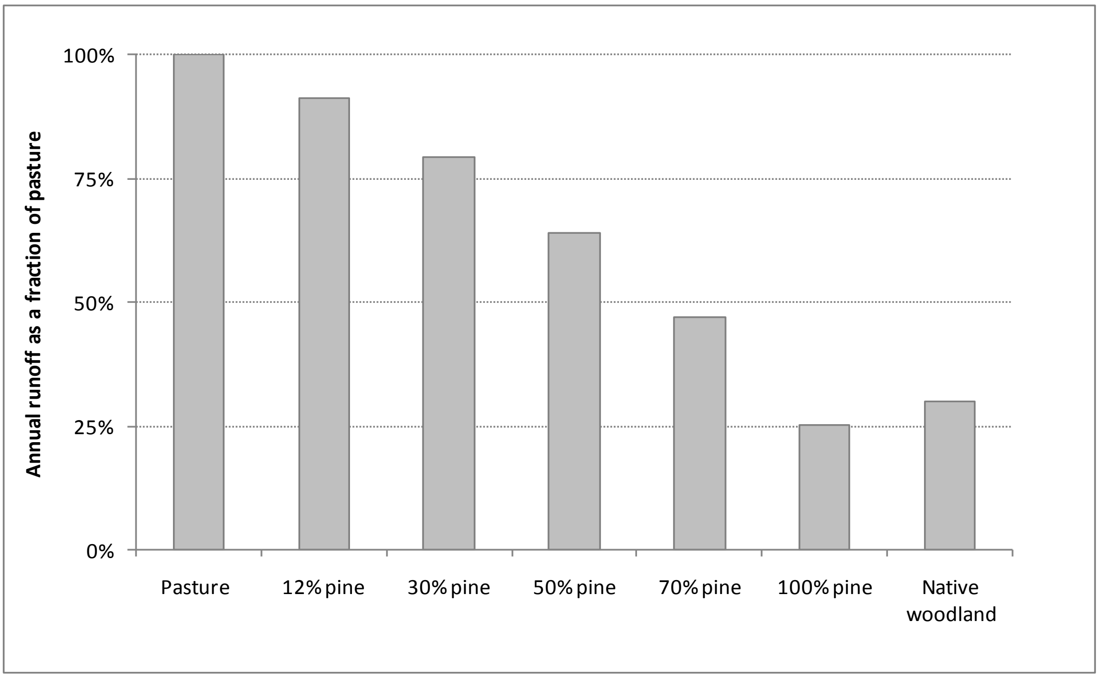


Figure 32. **Modelled annual median catchment runoff as a fraction of modelled pasture response for different 37-year land use scenarios. Pine scenarios are all with 20 m buffer.**

All scenarios with pine-forest planting fractions exceeding 30% of the catchment had a median annual runoff depth less than 75% of that for pasture (Fig. 32). Catchments with a 100% planting fraction (20 m buffer) and native vegetation produced a median runoff depth less than 30% of that for pasture (Fig. 32).

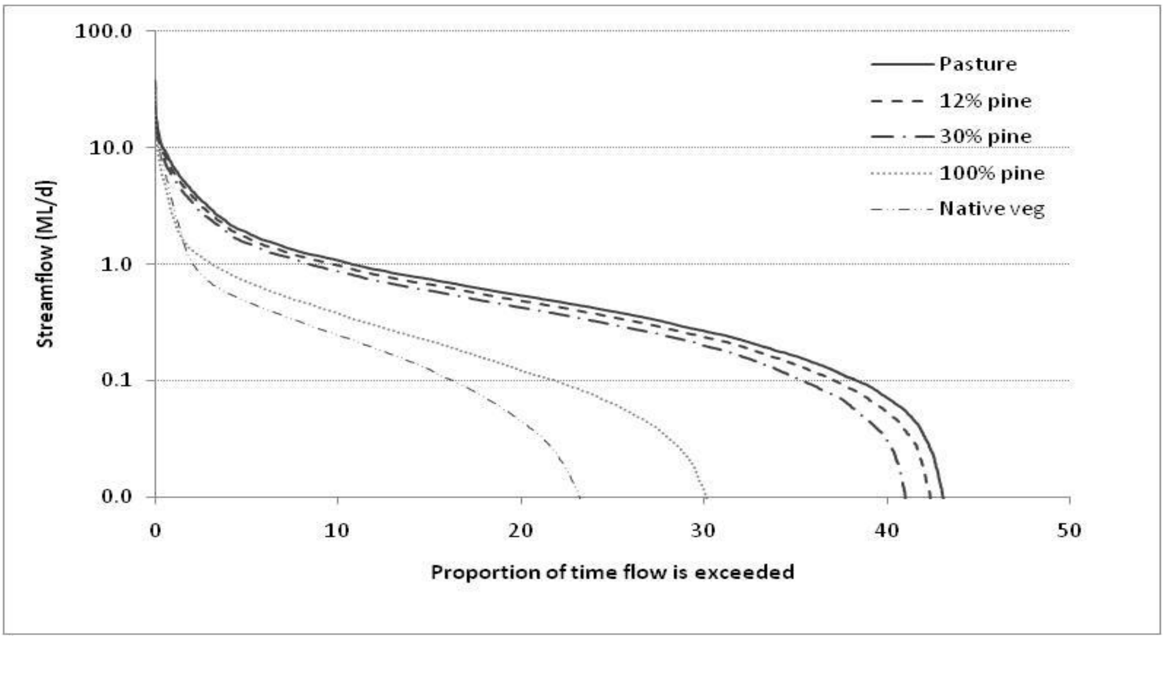


Figure 33. **Flow duration curves for selected land use scenarios modelled. Pine scenarios are 20 m buffer and full 37-year rotation.**

Under a pasture land use, the mean number of flow days per year for the 37-year scenario was 158, with the maximum continuous flow period being 169 days. Flow periods of 4-6 months duration are typical of low stream order intermittent watercourses in the Mount Lofty Ranges (e.g. Bestland *et al*. 2009).

With the exception of the native vegetation scenario, from which flow was not observed to occur in 2002, all simulations resulted in at least some stream flow every year. No scenario produced stream flow that persisted for greater than 45% of days over the modelled period (Fig. 33). As was found with a number of statistical descriptions of flow volume described above, pine-forest planting fraction had an inverse linear correlation with the number of days runoff occurred. In the case of closed native woodland, full understory native vegetation cover reduces the runoff frequency to around 24% of days and over a full rotation at 100% pine planting fraction flow is modelled to occur around 30% of days (Fig. 33).

Increasing buffer width increased runoff in a linear fashion, but the effects of buffer size on modelled runoff depended on the planting fraction for the modelled wetland of 6.3 ha (Fig. 34). For a 30% planting fraction, changing the buffer width from 10 to 20 m increases the resulting area by 1.4 ha. This is equivalent to 20% of the wetland area. This addition makes little difference to the observed runoff, increasing the median annual value by 1 ML/y, but having no notable effect on the mean value. Increasing the buffer from 10 m to 100 m represents an increase in buffer area of 14 times and results in an increase in median annual runoff of 6 ML, changing the value by only 4% (Table 8).

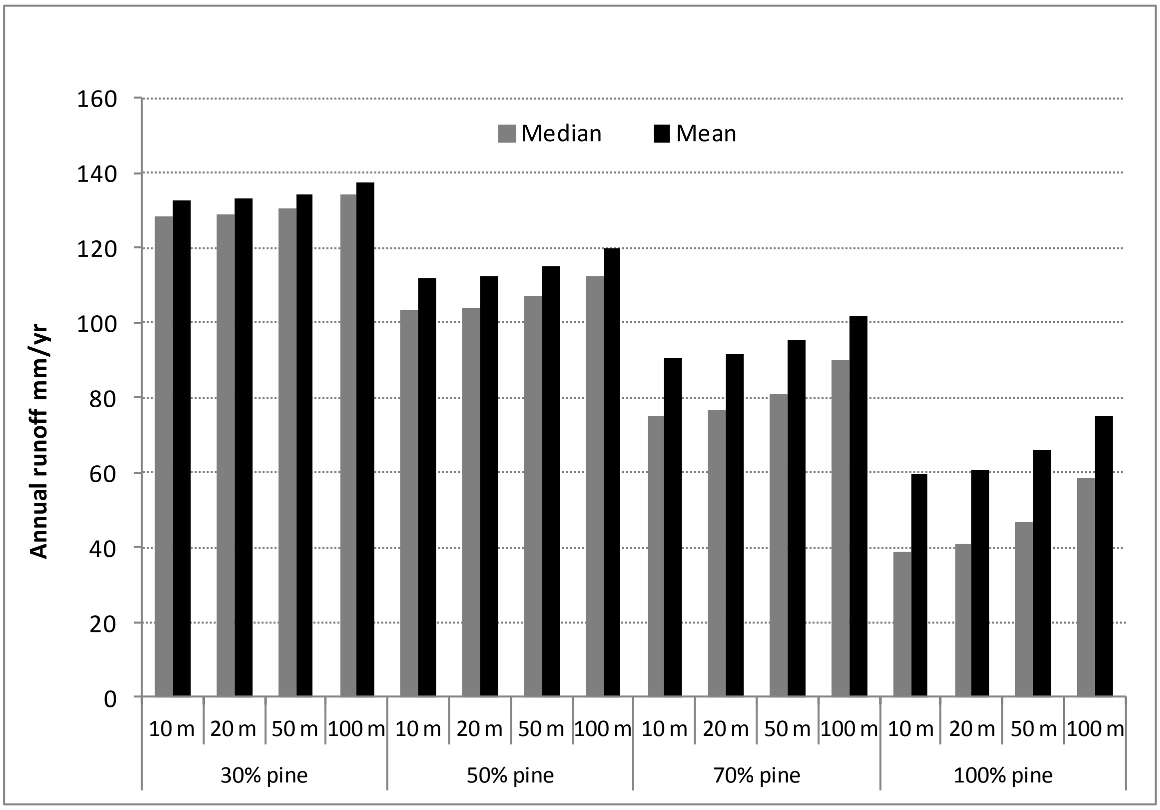


Figure 34. The effect of varying planting fraction and buffer width on mean and median annual runoff

At a 100% planting fraction, as buffer width is increased from 10 to 100 m (1.4 to 16.3 ha in terms of area), median annual runoff as a percentage of pasture runoff, changes from 24 to 36%. This represents a 50% increase in runoff, in comparison to the 30% planting fraction, where median runoff increases by around 5% for the same change in buffer area. Increasing buffer width from 20 to 50 m for a 100% pine planting fraction increases median annual plantation runoff from 40.8 to 46.9 ML, or 15% (Appendix A), only slightly below that predicted under native vegetation (48.6 ML).

Table 8. Selected scenario annual runoff statistics compared to that of pasture (native vegetation, 12% plantation and 30% plantation with four different buffer widths)

	Pasture	Native veg.	12% *20 m buffer*	30% *10 m buffer*	30% *20 m buffer*	30% *50 m buffer*	30% *100 m buffer*
Mean	164	53	151	133	133	134	137
Median	162	49	148	128	129	131	134
% of median	100%	30%	91%	79%	79%	80%	83%

Inverse linearity in modelled runoff response as a function of pine planting fraction is a consistent pattern in results supported by the literature. Published meta-analyses of paired catchment studies determining changes in runoff following deforestation have reported linear increases in yield following deforestation (Bosch and Hewlett 1982; Sahin and Hall 1996; Best *et al.* 2003). Bosch and Hewlett (1982) proposed a linear relationship where a 10% increase in pine planting would be predicted to reduce yield by 40 mm, while in this study the value was much lower (~13 mm). Linearity is consistent across scales, with Zhao *et al.* (2009) reporting on a similar relationship between catchment fraction cleared of forest and runoff across a range of scales from 1–10 000 km^2.

Greenwood and Cresswell (2007) undertook the only known analysis of catchment yield changes relating to afforestation from the region (the Mount Lofty Ranges), a case study at Burnt Out Creek about 60 km to the northeast of the study region. This work involved a statistical comparison of catchment yield data from a pine forest partially cleared following fire, with climatic variation controlled through the use of a paired site where land cover remained (more or less) constant. The authors concluded that planting a catchment at 100% *Pinus radiata* would likely reduce catchment water yield from that of a mixed land use by 70-100%. A use threshold of 85% was adopted for estimating the maximum impacts of forestry in state policy (GSA 2009).

This study has found a similar, though slightly lower, level of reduction (79%) between 100% pine and 100% pasture. This reduction is at the extreme end of predictions from Farley *et al.* (2005), who suggest data indicates a one third to three quarters stream flow reduction from afforestation as a general guide. Other authors have suggested reductions may be as high as 100% in lower rainfall areas, but no simulation led to zero runoff in this study (maximum annual reduction observed being 91% compared to pasture – Appendix A). Incorporating a 20 m buffer and modelling over a full plantation rotation (thereby including early period runoff prior to canopy closure), yield reduction compared with median annual pasture runoff is 75% for 100% pine forest.

Both the current study and the prior work by Greenwood and Cresswell (2009), report reductions for the Mount Lofty Ranges that are consistent with published findings from other parts of the world. Scott and Smith (1997) reported that the maximum reduction in runoff due to conversion of pasture to pine trees was between 85% and 100% based on observed data from paired catchment studies. Vertessy and Bessard (1999), Vertessy (2001) and Vertessy *et al.* (2002, 2003) reported that mean runoff from a grassed catchment could be expected to decline by 79% if planted to eucalypt forest and 100% if planted to pines. Theoretical forest and pasture runoff curves constructed by Bradford *et al.* (2001) and

Zhang *et al.* (2003) indicated that the difference between mean annual runoff from the pasture and forest ranged from 60–85% in areas with mean annual rainfall up to 1000 mm.

When the modelled and observed reductions for South Australian systems in terms of absolute depth of runoff are compared with the broader literature the results are equivocal. Farley *et al.* (2005) provide a similar estimate of changes to runoff to that estimated through this work. Values presented in Fig 6. (p 1570) of that meta-analysis suggest a comparable value for sub-1050 mm rainfall regions to the 103 mm difference in mean annual catchment yield observed between the pasture and pine forest scenarios in the current study. Other studies have shown greater reductions however.

Data presented (Zhao *et al.* 2009) suggests a similar level of increased yield to that proposed by Bosch and Hewlett (1982), where runoff was estimated to have increased by 350 mm for a 100% clearance of forest. The absolute magnitude as well as the proportional change observed between grassland and pine forest have however been found to be dependent upon a number of factors other than land use, most notably climate, especially rainfall.

Although from temperate zones, all studies of pine responses analysed in Bosch and Hewlett (1982) came from areas where mean annual rainfall exceeded 1200 mm. Best *et al.* (2003) re-analysed the data presented in Bosch and Hewlett (1982), re-scaling yield increases to incorporate rainfall variation between the study sites. Their Figure 2 (p. 9, Best *et al.* 2003) suggests a reduction for a 950 mm rainfall (equivalent to the study region) of a little below 200 mm—a value exceeding predicted total runoff under pasture in the present study. The value from Best *et al.* (2003) is in accord with data presented in Sahin and Hall (1996), who in an analysis of 145 studies, concluded that a 10% decrease in conifer plantation would increase yield by between 20 and 25 mm. This value is around twice the rate of increase per 10% planting estimated from this study of 13 mm for a zero buffer or 10 mm for a 20 m buffer.

5.3.2. SEASONAL VARIATION IN MODELLED SCENARIOS

As observed above, stream flow regimes for all modelled scenarios were intermittent. Modelled runoff from wetlands under pasture generally commenced during May and continued through until January at the latest (Fig. 35). Stream flow events were only observed to occur in the month of February for the pasture and 12% planting scenarios and only on one occasion (1993).

Pasture and the lower planting fractions (12% and 30%) all saw streamflow regularly commence as early as May, although June and July were less variable as indicated by the coefficient of variation (Fig. 35). Flow most years ceased in November or December. The streamflow season for the low planting scenarios encompasses up to eight months of the year, but most commonly was restricted to the six months June–November. The season also comprises a number of discrete flow events at the early and later end of the season rather than continuous flow.

Stream flow volumes for all land uses were highest during July and August. Mean values were clearly higher for August (Fig. 35), suggesting larger stream flow events are perhaps most likely during this month. Median values during flow season months were highest in July, suggesting this is the most consistent month (Fig. 35).

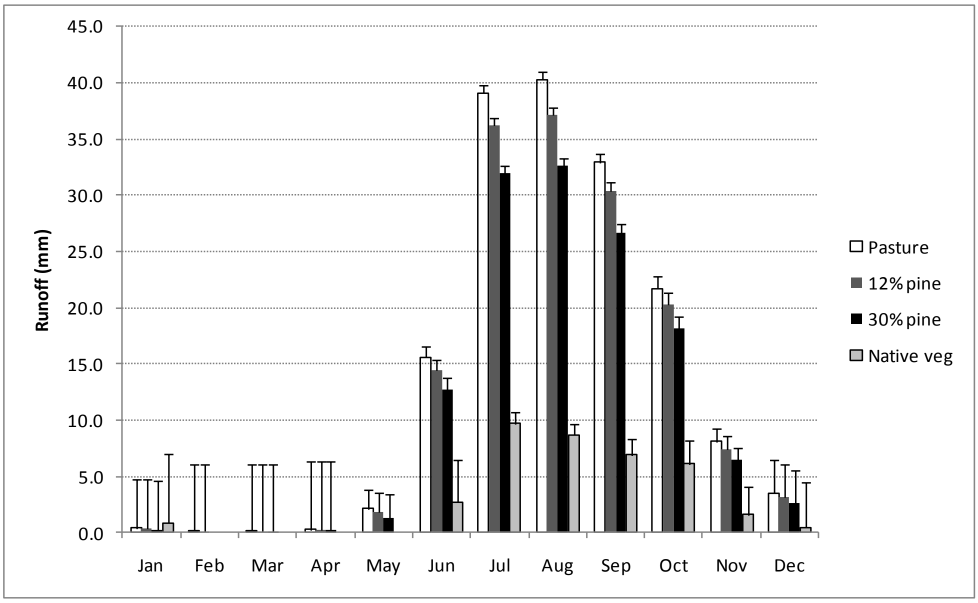


Figure 35. Modelled mean monthly stream flow for various scenarios. Error bars represent coefficient of variation. All forest buffer widths 20 m. Native veg is closed stringybark woodland.

Mean monthly stream flow volumes for flow season months over the full plantation cycle are reduced across all months, with the reduction closely correlated with planting fraction (Fig. 35, Table 9). A planting fraction of 12% pine results in a decrease in mean monthly stream flow volume of between 7 and 16% during the April–December period (Table 9). The reduction range increases to 17–36% for a 30% planting fraction and continues to increase with planting proportion to between 62–85% under a fully-planted scenario (Table 9). The greatest impacts for all planting fractions are on early and late flow season months of April, May and December. The lowest deviation for all scenarios was during October (Table 9).

Table 9. Percentage reduction in mean monthly streamflow statistics relative to that observed for pasture during the streamflow season over a full forest rotation

Scenario	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
12% pine	11%	16%	7%	7%	8%	8%	7%	8%	10%
30% pine	25%	36%	18%	18%	19%	19%	17%	20%	25%
30%, 50 m Buffer	24%	35%	18%	17%	18%	18%	16%	19%	24%
30%, 100 m Buffer	21%	32%	16%	16%	16%	16%	14%	17%	22%
50% pine	41%	55%	31%	30%	32%	31%	28%	33%	41%
70% pine	57%	69%	45%	43%	45%	44%	39%	46%	57%
100% pine	82%	85%	66%	62%	64%	63%	55%	64%	79%
100%, 50 m Buffer	78%	84%	63%	59%	61%	60%	52%	61%	75%
100%, 100 m Buffer	70%	80%	56%	53%	55%	54%	47%	56%	69%
Native vegetation	100%	100%	83%	75%	78%	79%	72%	81%	87%

- Months not shown had zero mean stream flow value
- Unless specified otherwise, all pine plantation scenarios had a buffer width of 20 m

Monthly streamflow reductions compared with pasture are mostly little affected by the buffer width (Fig. 36). May and October values which are virtually unaffected, but for other months each buffer size increase also decreases the observed reduction when compared with monthly median pasture streamflow (Fig. 36).

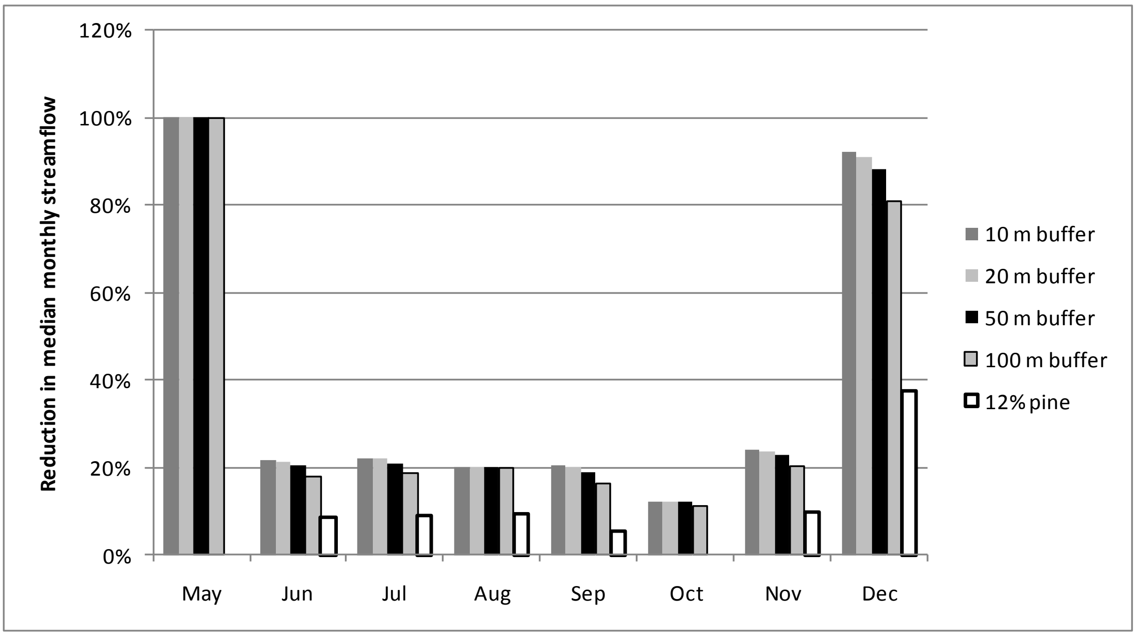


Figure 36. Reduction in median monthly streamflow compared to pasture for a 30% planting fraction, as a function of buffer width (12% planting fraction with 20 m buffer shown for comparison)

The literature has few examples of the effects of forestry on the seasonality of stream flow (Best *et al.* 2003). Farley *et al.* (2005) report the generally held prediction that proportional impacts on stream flow will be greatest in drier climates and present data that suggest the greatest impact is on dry season runoff. Best *et al.* (2003) also present some data on loss of low flows, reported at up to 100% for Redhill catchment in Victoria. Modelled dry-season runoff in the study region is zero or at least highly irregular. The greatest proportional impact on modelled monthly flow occurs in early and late season stream flow in all low flow season months (April, May and December) (Table 9; Fig. 35). This is consistent with the predictions presented in Farley *et al.* (2005) that lower flows are most heavily impacted.

5.4. PLANT FUNCTIONAL GROUP MODELLING

The means by which perched wetland ecology might change as a result of different land use scenarios was assessed through modelling the vegetation community. Changes to wetland hydrology were strongly linear in nature, meaning a proportional increase in forested area would result in a proportional decrease in the amount of water available to support wetland plant communities. The aim of this assessment was to determine at what point on this continuum wetland communities might be expected to make a recognisable transition with all other variables held constant.

Probabilities from the six functional group models for selected scenarios, along with two estimates for each of the three field sites (indicative of the range of conditions at each), are shown in Table 10. Functional group Tdamp (terrestrial damp), is the most broadly suited to water regimes across all scenarios with a mean probability of 0.71. Aftw (amphibious fluctuation tolerator woody growth habit—most commonly prickly or woolly tea-tree in this region) and Afte (amphibious fluctuation tolerator emergent—typically sedge species) are also well suited to the range of conditions, with mean probabilities across modelled scenarios of 0.58 and 0.55 respectively. This is broadly consistent with the analysis in Section 3.2 for perched wetland plants based on presence absence data and that of Appendix H using the field data at transect level to characterise the difference between field sites. Seasonal perched wetlands apparently provide habitat most suited to semi-aquatic wetland vegetation.

In contrast, SE (submerged emergent – preference for permanent surface water or permanently saturated root zone) is poorly suited to scenario water regimes; no scenario has a modelled probability of SE exceeding 0.5 and the mean probability across all scenarios for the PFG is 0.29. Only Tdry (terrestrial dryland species) with a mean probability of 0.28 is less well suited to scenario water regimes overall. This finding is also consistent with the lower proportional occurrence of SE species in the perched wetlands species data from Harding (2005) in Section 3.2, suggesting many of those sites were seasonal in nature. Despite this relatively low occurrence, submerged species are clearly a component of many perched wetlands (Sect. 3.2), including those with a seasonal hydrology (Appendix H). Micro-topographical variation has been shown to exert considerable influence on spatial patterns in wetland floristic expression (Bruland and Richardson 2005) and no doubt plays a role. It is probable that where present in seasonal perched wetlands, submerged functional groups will be spatially restricted to areas with longer duration periods of saturation such as watercourses or small areas of permanent saturation associated with springs. Such species are indicative of wetlands (and areas within them) that remain wet for long periods. These are also likely to be the first groups lost from seasonal perched systems under enduring water stress.

For modelled scenarios, a wetland catchment under 100% pasture returns the highest probability for all functional groups with a known dependence on saturated conditions. Pasture also predicts the lowest probability for Tdry, which has high probabilities only for full planting fractions with zero or 20 m buffers and native forest. Generally probabilities for the amphibious and submerged functional groups exhibit the reverse trend to Tdry, decreasing with increasing planting fraction. Survey data from this study also found that terrestrial species were characteristic of the shorter duration watertable persistence at Foggy Farm (Appendix H).

Table 10. Plant functional group probability of presence: above dashed line are modelled scenarios, below dashed line modelled probabilities for field site hydrographs (decreasing wetness requirement from left to right)

Scenario	SE	Afrp	Afte	Aftw	Tdamp	Tdry
Pasture	0.45	0.56	0.75	0.75	0.86	0.13
20B12	0.42	0.54	0.73	0.73	0.85	0.15
20B30	0.39	0.50	0.68	0.71	0.82	0.16
20B50	0.31	0.46	0.62	0.63	0.77	0.22
20B70	0.29	0.42	0.56	0.61	0.73	0.23
0B100	0.09	0.24	0.25	0.31	0.44	0.56
20B100	0.12	0.27	0.30	0.39	0.50	0.47
Native	0.10	0.29	0.34	0.33	0.54	0.53
50B100	0.16	0.33	0.42	0.46	0.62	0.38
100B100	0.21	0.35	0.44	0.53	0.63	0.32
50B30	0.39	0.50	0.68	0.71	0.82	0.16
100B30	0.39	0.51	0.69	0.71	0.82	0.16
50B50	0.33	0.46	0.63	0.65	0.78	0.20
100B50	0.35	0.47	0.64	0.67	0.79	0.19
SR_US	0.20	0.47	0.64	0.52	0.79	0.32
SR_DS	0.20	0.34	0.42	0.51	0.62	0.33
WS_Main	1.00	0.83	0.96	1.00	0.98	0.00
WS_Trib	0.57	0.30	0.35	0.82	0.55	0.09
FF02	0.04	0.24	0.25	0.19	0.44	0.72
FF04	0.02	0.13	0.10	0.10	0.22	0.86

Buffer areas have an effect upon modelled probabilities which is dependent upon planting fraction. Increasing a 30% planting fraction buffer width from 20 to 100 m has no effect on the modelled probability of whether any functional group increases or decreases. For a 100% planted catchment the same increase raises the probability of submerged and amphibious groups being present by between 25 and 73%. A concurrent effect of this change in buffer area is the decrease in Tdry modelled probability of over 30%. For a planting fraction of 50 or 70% increases in aquatic plant probabilities are observed as buffer areas increase, but are in the range of 3–13%.

The highest probabilities for the observed data are those for the Wither Swamp (Table 10) and the perennial channel site (WS_Main see Fig. 59 for location) predicts the presence of all amphibious and submerged PFG at probabilities exceeding 0.8. Tdry has a probability of zero at this site. This is in accord with observations at the site that found the perennial near surface saturation supported the highest species richness, apparently due to the provision of habitat suited to a broader range of plant functional groups (Appendix H). Springs Road sites (see Fig. 57 – Site SR_US downstream of the pasture area is the wettest and SR_DS at mid-wetland is the drier of the two locations sampled at this site) are intermediate in probabilities between the Wither Swamp and Foggy Farm predictions.

Comparisons between observed field data and the analogous model prediction are not appropriate for Springs Road native forest, as the wetland was subject to additional runoff inputs from pasture and cannot be compared to the native forest modelled probabilities. The confined groundwater flow system generating perennial surface saturation at the Wither Swamp wetland also means comparison with the modelled pasture scenario is inappropriate.

For Foggy Farm, predicted probabilities in Table 10 can be verified against observed monitoring data in Table 20 (Appendix H), though it should be noted that predicted probabilities are not directly equivalent to proportional vegetation cover. Positive correlations are apparent, though this would be expected given the observed data were used in model construction. The two Foggy Farm wetland sites predict low probabilities for all PFG preferring wetter conditions and Tdry species with probabilities exceeding 0.7. Monitoring site FF04 (see Fig. 58) is located at the upstream end of the wetland and would represent conditions in a gully adjacent to, but outside the wetland itself. FF02 is located at the most downstream end of the wetland and represents the wettest conditions at the site that is still within the extant tea-tree zone in the wetland.

5.4.1. CLASSIFICATION OF MODELLED PLANT COMMUNITIES

Classification to investigate likely groupings of the scenarios was undertaken using UPGMA ('unweighted pair group method with arithmetic mean') clustering (see Sect. 4.2.4). Figure 37 presents selected scenarios classified on the basis of the predicted plant functional group probabilities. Two major groupings are evident, representing fully planted and partially cleared catchments. Four classes are created by a cut-off threshold at height (y axis units – Bray-Curtis dissimilarity) 0.06 and these are indicated by the grey rectangles in Figure 37.

Groups number left to right and comprise:

1. planting fractions of 30% or below
2. intermediate planting fractions of 50 and 70%
3. native vegetation and 100% pine scenarios with zero or 20 m width buffers
4. 100% planting fraction with 50 and 100 m width buffer areas.

Grouping accounts for around 60% of the variation and the differences in probabilities between these groups are greater than the within group variation at statistically significantly levels ($p < 0.01$) based on permutational multivariate analysis of variance (Oksanen 2010).

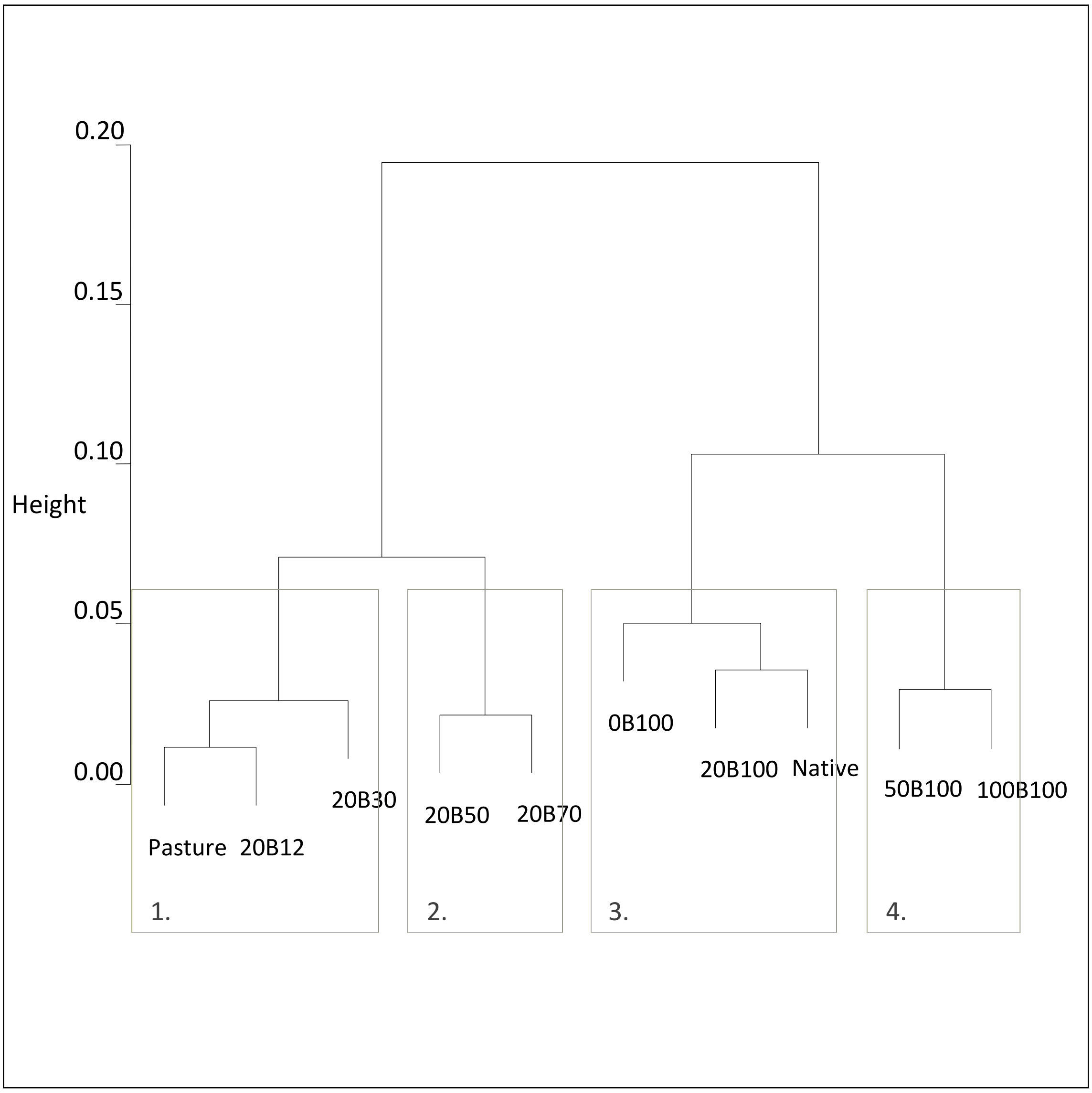


Figure 37. UPGMA hierarchical classification of arcsine transformed Bray-Curtis distance matrix for selected modelled scenarios functional group probabilities. Groupings in rectangular boxes are statistically significant (permutational multivariate ANOVA; $p < 0.01$; $R^2 = 0.61$).

Relative probabilities for different PFG exhibit consistent patterns within the four classes, with the exception of Tdry, Aftw and Afte. In pasture and 12% planting the latter two are of the same probability, while in higher planting fractions Aftw is of consistently higher probability. Tdry is the highest probability in group 3, but lowest in groups 1 and 2.

Not shown in the figure to increase clarity are the buffer scenarios for plantings other than 100%. In all cases, 20, 50 and 100 m buffers clustered together in sub-groups. In each case either the 20 m or the 100 m buffer scenario was a singleton chained to the other two with a very small height distance between the three predictions. No increase in buffer width modelled led to a given planting fraction grouping with a lower planting fraction.

In the Figure 37 dendrogram the relative height between groups 3 and 4 indicates a considerable difference in community structure for a fully forested catchment where a larger buffer is imposed. Differences between pasture and a 12% planting fraction are the least of any two scenarios, although the grouping of this sub-group with the 30% planting is of a greater difference than predicted between a 50 and 70% planting fraction. Native forest probabilities are predicted to be very similar to a 100% pine plantation with a 20 m buffer, a result inconsistent with the observed data (Appendix H).

Reductions in the duration of watertable persistence at the wetland surface would result in ecological effects that increase with planting fraction. Modelling suggests that at the broadest level of community assembly, a wetland within a fully forested system (whether native or plantation) will be very different compositionally from a catchment with at least 30% of pasture.

The importance of groundwater dynamics in establishing the composition and zonation of wetland plant communities is well established in the scientific literature (e.g. van der Valk 1981; Rea and Ganf 1994; Van Splunder 1995; Ganf *et al.* 2000; Keddy and Fraser 2000; van der Valk 2005; Smith and Brock 2007; Watt *et al.* 2007). Most published evidence suggests the most influential variables relate to the period of time that wetland sediments remain inundated and the depth–duration dynamics of this period (Froend and McComb 1994; Ganf *et al.* 1999; Ganf *et al.* 2000; Nicol *et al.* 2003). However the depth scenarios typically examined greatly exceed anything likely in perched Fleurieu wetlands. At the drier end of the water-availability continuum, the days per year of exposure (i.e. watertable below the surface) and the maximum depth below ground to which the watertable falls are critical variables (Froend and McComb 1994). This latter consideration is clearly most relevant to the seasonal perched wetlands modelled, which rarely inundate to more than a few centimetres.

For the perched wetlands study the period of surface saturation appears particularly influential in structuring communities. Variables best describing functional group presence or absence in modelling work were derived from the estimated annual duration of the watertable at near the wetland surface. This corresponds to saturation of the shallow loam layer overlying the clay subsoil, where root development is not constrained (see Sect. 5.2). Although quadratic terms were included in saturated models where justified on theoretical grounds, no terms increased model likelihood (or predictive power) sufficiently to warrant inclusion in the final model formulation. This provides an insight into the nature of seasonal perched wetland vegetation in that water regimes are unlikely to be limiting to any functional group at the wetter end of the water availability continuum for modelled water regimes. As such assessments for hydrological impact on these systems may need to focus on changes to relative cover or abundance of different plant groups.

Functional groups characterising fully-forested seasonal perched wetlands are basically terrestrial, although Aftw species may also persist within the community, though field evidence as to whether this pattern would be expected in a native woodland is equivocal. Data collected at Springs Road wetland indicates the wetland community is closer in affinity to Wither Swamp than Foggy Farm.

Were a change in water availability for a perched wetland to occur, the vegetation community may take considerable time to adapt. The deeper root architecture and longer life span of woody wetland species may allow such species to persist during drought periods (Casper and Jackson 1997), even if regeneration requirements are not met—an example of an ecological lag. Owing to their ability to access water from deeper in the soil profile, it is reasonable to predict that these may be the last recognisable wetland functional group to be lost from a water-stressed wetland. The Foggy Farm wetland would appear to be in just such a successional shift in vegetation, based on the low representation of non-woody wetland plants and relatively high proportion of terrestrial species (Appendix H).

Any site where Aftw was the only amphibious functional group present would suggest the wetland may have been under persistent water related stress. In such cases comparing seed bank diversity with existing plant community expression through germination trials can also provide a means to determine water regime driven succession. If species from functional groups not currently present in the wetland and of a higher water requirement were to be successfully germinated, this would suggest a shift in water supply regime has occurred (van der Valk 1981).

Small differences between modelled community composition at 30% planting fraction to those below 12% are evident in the results, but in reality may be difficult to measure. If a wetland catchment originally under pasture was planted to 30% forest it is possible that over time some functional groups (ultimately meaning plant species) may shift in relative abundance. Recognising that functional groups themselves contain variability, ultimately this raises the potential that a marginal species at the high water requirement end of the spectrum may eventually be lost from the wetland. Over time any reduced suitability of habitat would be expected to decrease the competitive ability of the functional group, decreasing the probability that the disadvantaged functional group will be found within the system. Absolute thresholds for such transitions are currently unknown.

The predicted plant community under the < 70% planting fraction scenarios were not greatly influenced by the size of the buffer zone over the range modelled, but buffers appear highly influential on 100% planting scenarios. This analysis suggests that communities within fully planted catchments where the buffer area was less than half of the wetland size itself, would be expected to be recognisably different from wetlands where buffer areas were at least 1.2 or more times the size of the wetland. For modelled wetland dimensions this corresponds to buffer widths of 20 m or less and buffers 50 m or greater. Quite different communities are predicted for small or no buffer than for the larger buffer areas, including a good probability of supporting Aftw species and also possibly Afte species. Changes in relative abundance would be anticipated based on modelled results and changes in composition may also occur, depending perhaps on the state of local seed banks or the proximity of a source population for propagules.

6. SUMMARY AND CONCLUSIONS

6.1. PERCHED WETLAND HYDROLOGICAL PROCESSES

A high degree of variability in groundwater and surface water processes was observed across the three catchments used for this study (Sect. 5.1; Figs 21–28) and these differences were clearly reflected in wetland vegetation at functional group level (Appendix H). The three conceptual soil water movement pathways identified offer a possible explanation for the observed differences. Surface water runoff in Mount Lofty Ranges catchments with texture contrast soils has been observed to commence only when rainfall exceeds A and B horizon throughflow demand or infiltration capacity (Chittleborough 1992; Bestland 2009), however this is far from consistent. Prior work has demonstrated that the relative importance of these pathways for catchment water movement differs considerably between sites (Smettem *et al.* 1991; Leaney *et al.* 1993; Fleming and Cox 2001; Cox *et al.* 2002; Bestland *et al.* 2009). Understanding the relative importance of each of these processes and the manner in which water resource development will affect water storage and movement in perched wetlands is critical to future management, but far from trivial and requires additional investigation.

For example, farm dams intercept overland flow and throughflow in the A horizon, are less likely to intercept B horizon throughflow and will not affect a confined flow system. The latter two processes are more likely to be critical to persistent saturation in wetlands. Plantation or farm forestry would potentially affect B horizon throughflow, especially if subsoil ripping occurs prior to planting. Any effects on overland flow and A horizon throughflow depend in part on how infiltration to B horizons is affected as this store tends to fill preferentially in texture contrast soils of the region (Chittleborough 1992). If plantations were located over the recharge or discharge zones for a pathway 3 confined flow system, commercial or possibly even native woodland restoration could exert considerable influence on wetland hydrology, perhaps disproportionate to planting area related changes as suggested by this study. The effects of forestry on overland flow are probably quite well represented in this study and existing surface water models appear capable of re-producing this.

The ‘numerous’ permanent springs adjacent to plateau surfaces described in Campana and Wilson (1953), are considered likely to represent B horizon throughflow discharges (pathway 2 in Fig. 23). Permanently flowing springs do not appear to be common in the contemporary landscape, although perched wetlands certainly are and in the Parawa region are reported to have increased in area since clearance mid last century (P.Filsel 2009, pers. comm.).

Wetlands adjacent to remnant plateaux surfaces were described as the area supporting springs in Campana and Wilson (1953). Planting and especially ripping headwater gullies above wetlands may interrupt pathways that would move water via subsoils to wetlands. The Foggy Farm and Springs Road sites are candidate locations to investigate this question, especially if an additional seasonal wetland under pasture (that is lacking the additional flow component present at the Wither Swamp) was available for comparison.

Historical references aside, it appears from water balance modelling and field observation that perched wetlands reliant on overland flow and throughflow via A and B horizon pathways only will be limited to seasonal saturation across much of the wetland. Seasonal wetlands are among the most vulnerable to human impact (Holland *et al.* 1995) and seasonal and supra-seasonal drought provides regeneration opportunities for terrestrial species to become established (Casanova and Brock 2000). The short depth and duration hydro-period makes seasonal perched systems susceptible to terrestrial invasion (Sect. 5.2, Appendix H). The plant community of water stressed seasonal perched wetlands can be expected to shift to a higher dominance of terrestrial natives or weed species, especially as most Fleurieu wetlands are highly integrated among agricultural lands.

Perched Fleurieu wetlands with permanent saturation near the wetland surface, if not perennial streamflow, are highly valuable aquatic refugia, supporting increased species and functional group diversity (Appendix H). Ecological values of such systems extend beyond the vegetation community. For example the climbing galaxid, a small native fish, has been observed within the watercourse at the Wither Swamp. Until additional understanding is available, any such sites identified should ideally be viewed as vulnerable to land use change.

Determining the prevalence of perched wetlands with permanent saturation and streamflow and understanding how land use and water resource development affects wetland hydrology are viewed as priority knowledge gaps. This understanding is critical for water allocation, conservation and restoration planning purposes in the region.

6.2. SCENARIO MODELLING

Modelled changes to land use in seasonal perched wetland catchments exerts considerable influence on observed stream flow dynamics. Modelling suggests that relative influences are quite predictable, with most statistical measures of hydrology exhibiting a strong inverse linear relationship with the planting fraction of the catchment. Generally the proportionality of this relationship is less than 1, meaning a 1% increase in forestry reduces the magnitude of most hydrological statistics by a slightly lesser proportion. For example, mean annual runoff decreased by around 0.63% for every 1.0% increase in pine forest planting fraction (Fig. 31). In terms of the effects on catchment yield expressed as a depth of runoff, a 0.63% decrease will result in a reduction of around 13 mm for every 10% increase in planting fraction.

The Statewide Policy Framework (GSA 2009) adopts the 25% sustainable limit and the position that plantation forests will reduce runoff by 85% within the area that is planted. For the purposes of this study, it is therefore considered that the 25% sustainable limit equates to a 30% planting fraction. This is also supported by the work of Greenwood and Cresswell (2007). The Statewide Policy Framework (GSA 2009) also adopts the position of 20m buffer widths for surface water resources.

This study has confirmed that, for the study area, a 30% planting along with a 20 m buffer achieved the policy aims of a 25% sustainable limit, estimating the median annual stream flow as being 79% (i.e. 21% usage) of median annual runoff from pasture. Thus a 30% planting fraction appears to build in a small tolerance margin, which is important given uncertainties around wetland to catchment ratios. Modelled scenarios where the planting fraction exceeded 30% (50, 70 and 100%) were found to use in excess of 25% of the median runoff predicted for pasture irrespective of buffer size. At an annual time step, reductions in runoff from the 30% scenario exceeded 25% of the predicted pasture runoff on three occasions over the 37 year simulated rotation. The maximum reduction observed in any given year did not exceed 27%.

Given the predictable relationship observed, it is not surprising that runoff estimates for the 12% pine planting fraction were very similar to those obtained for the pasture scenario. The 30% pine scenario, while being further removed from the pasture response, typically returned statistics with variations of around 25% or less. No pronounced change in flow magnitude, duration, or seasonal timing were identified at these low planting fractions; statistics were simply reduced in magnitude at levels reflecting the amount of catchment planted. Daily flow duration curves (Fig. 33) demonstrate the similarity of the response across the pasture, 12% and 30% planting scenarios.

In terms of their impact on catchment hydrology, buffer widths of up to 100 m (2.5 times the area of the wetland itself) were found to make a limited difference to modelled runoff statistics under low planting fractions. For example, even a buffer of 100 m width and representing an area more than twice that of the wetland itself did not substantially modify annual runoff in the 30% planting fraction. Runoff as a percentage of the median pasture response was increased in this case by only 79 to 83%, whereas changing the buffer distance was highly influential on wetland hydrology in higher planting fractions, especially for fully planted catchments.

Imposing a uniform buffer width for all wetlands will not result in a uniform level of protection, owing to variations in catchment to wetland area ratios. Determining buffer widths based on some multiple of the wetland and catchment areas is recommended as a more accurate and site specific method for perched systems in particular.

The use of functional groups to analyse vegetation community has provided some clear indications of the manner in which the vegetation community within seasonal and perennial perched wetlands differs and also how water stress may manifest in the plant community. Ecological data analysis (Sect. 3.2; Appendix H) and modelling (Sect. 5.4) suggest a number of generalisations around the likely community expression. In terms of broad plant functional group composition, low stream order seasonal perched wetlands are likely to be dominated by amphibious and inundation-tolerant terrestrial vegetation, though functional groups of higher water requirement may well be present in low proportions. Perched systems with permanent saturation at the surface provide good habitat for all functional groups, semi-aquatic and true aquatics.

Determining hydrological conditions from the plant community and seed bank is a well established approach (van der Valk 1981; van der Valk 2005). The magnitude of seasonal fluctuations has been used to categorise prairie pothole wetlands in the United States into broad classes and as a basis to develop models capable of predicting vegetation successional changes at any point of a wetting and drying cycle (van der Valk 2005). The use of vegetation models in establishing groundwater dynamics based on observed vegetation communities has been successful for comparable systems of fluctuating hydro-period (Kennedy *et al.* 2006). Such tools are powerful aids to management, particularly for prediction in the face of management challenges such as climate change and could readily be developed for these systems.

6.3. MANAGEMENT CONSIDERATIONS

6.3.1. APPLICATION OF FINDINGS

The short duration of monitoring record upon which findings are based means that this work must be considered to be preliminary. Estimates and modelled outputs remain unverified over time and are therefore of insufficient rigour to be applied directly for policy development or water resource planning. The project has highlighted a range of areas where existing conceptual understanding was incomplete and has improved general understanding of the behaviour of perched wetland hydrology and vegetation. Modelling undertaken was essentially a trial of the methods employed and further development is required to improve certainties around both wetland and catchment hydrology and wetland plant responses. Section 7 discusses some future research opportunities.

The broad range of observed conditions across only three perched wetland systems demonstrates the natural variability and how little is still understood of the interaction between land management, hydrology and ecology. A much greater research effort is required if sufficient general understanding is to be generated to provide effective management across the large number of perched wetlands in the region. There is no way of knowing whether the full spectrum of process variability has been observed through this project. However it is likely that future studies will identify additional processes that will need to be incorporated into conceptual models.

From the perched wetland management perspective, it appears critical that understanding of the presence and relative importance of overland flow and the three identified soil water movement pathways is developed. Land use and management, including water resource development, will likely affect each pathway in different ways. Causal influences on wetland condition are yet to be fully understood and this is an important management and water resource policy consideration.

Arguably the most notable observation of this study is the recognition of an additional soil water movement pathway contributing to perched wetland hydrology in (at least) the southern Fleurieu region. Soil water movement via pathway 3 (Sect. 5.2; Fig. 23) was only observed at the Wither Swamp site, but based on anecdotal reports other permanently saturated wetlands exist in the region and these may be dependent upon similar local confined flow systems.

Identifying the position of confined flow systems within perched wetland catchments may be essential for management purposes. If only a small proportion of the wetland catchment was underlain by such a flow system, significant land use change on the recharge and discharge areas could be expected to have a disproportionately high and possibly transformative, impact on wetland hydro-ecology. This raises the potential that even only planting 30% of a catchment would fundamentally alter the hydrology of the wetland if it happened to be the 30% of catchment area underlain by a pathway 3 system supporting perennial saturation. Future management will rely on additional investigation to inform appropriate policy development.

The existing Statewide Policy Framework (GSA 2009) is broadly supported by this study in terms of desired outcomes for water resource usage. This study has however identified that a standard buffer policy, which enforces a single planting threshold and buffer irrespective of wetland and catchment characteristics, may not necessarily achieve the desired outcomes for all wetlands. This is because the relative size of the wetland and its contributing catchment dictates what planting fraction it can accommodate without compromising wetland ecology (Casanova and Zhang 2007). This study has only modelled one possible combination of wetland volume and catchment size and in reality both of these measures occur on a continuum. The ratio used in modelling of around 7% was based on field sites, but this may be lower than typically observed. A random sample of perched wetlands data had a mean proportion of wetland to catchment area over 10%. This would mean impacts on wetland hydrology may be underestimated in this study. For a large wetland in a small catchment, a 30% planting fraction may impact more heavily than predicted. In the reverse set of circumstances, it is possible that some increase above the 30% threshold may be possible without compromising the 25% rule. Further investigation of more flexible policy options that reflect buffer distances based on wetland demand and catchment area are proposed.

6.3.2. UNCERTAINTIES

6.3.2.1. Hydrological modelling

The major uncertainties associated with the hydrological model are related to the period of data available for model calibration and the quality of the available data. The nature of the hydrology of the catchments combined with the timing of the project have effectively meant that only a single flow year was available for model calibration. This is extremely limiting, as it means catchment response has only been observed under a single set of conditions. In addition, the first year of the study (during which time monitoring was being deployed) was a serious drought year. This raises the possibility that catchment response observed was only typical of sites recovering from an extended drought period. The calibrated year (2009) received very close to long term mean annual rainfall (962 mm), but ideally sites should be maintained for a period of five years in order to gain additional certainty in the model parameter values.

The greatest uncertainty around the calibration relates to the native vegetation study catchment. This catchment had an area of around 15% of the catchment under pasture at the upstream end of the wetland. As this input could not be gauged directly, it was necessary to estimate this contribution through modelling. The woodland model node was calibrated to the difference between gauged total runoff and modelling pasture runoff, but if pasture response was overestimated, this would reduce the expected runoff from the woodland.

Field observation from early in the 2010 flow season at the native woodland study catchment indicated that initial runoff for the year had been generated from within native vegetation rather than the upstream pasture. The downstream gauge was recording stream flow in June 2010 and had filled a moderate size farm dam downstream of the wetland catchment. Dams on the boundary between the upstream pasture area and the native forest had not at this time reached full supply, indicating the area had made no contribution to the observed stream flow leaving the catchment. This increased uncertainty in relation to estimates for the native vegetation scenario in this report, which are likely to be lower than this land cover yields in reality. As with all of the modelling work, estimates of runoff under remnant forest can only be improved with a longer monitoring period. Native vegetation response is the least certain of all land uses modelled and requires additional work to improve the estimates.

Owing to the short monitoring record and intermittent flow regimes, only limited manual stream flow gaugings have been possible, meaning stage-discharge relationships are only theoretical over large flow ranges. Extending the period of operation of the sites beyond project completion will also allow for this to occur and increase the value of the stream flow record at the site.

The use of the lumped conceptual model is a limitation in terms of the spatial representation of catchment land use. A spatially distributed modelling platform could be considered for future follow up studies. Now that general catchment characteristics are better understood, such work is greatly facilitated. The modelling platform also imposed limitations on the precision with which watertable dynamics could be modelled. While the general behaviour was close to that observed, the surface dynamic of the wetland is complex and the approximation employed is limiting owing to the very different hydraulic properties.

In terms of the technical modelling approach, the major uncertainty was in simulating a full forest rotation without any suitable data for calibrating parameter sets. Data from the Burnt Out Creek site were used to calibrate the early growth phase following clear fell. The time for the forest to reach canopy closure was adapted from Greenwood and Cresswell (2007), comprising three years of the clear felled data and a further three years for the forest to reach canopy closure (Don McGuire (ForestrySA) 2010, pers. comm.). This period allows an extra year to compensate for the drought year in that dataset. The approach still slightly over-estimates the additional runoff during the periodic thinning operations over a planting rotation. However, no data were available to test such a simulation phase.

In the absence of any site collected data to confirm the level of increase during such periods and against which to calibrate a runoff model parameter set, these values can only be considered indicative. High quality monitoring data over a full forest rotation, preferably replicated and with a paired control catchment, is recommended to improve current capacity to model such variation with confidence. Only in this manner will the full range of influences on catchment hydrology over forest rotations be able to be incorporated into planning in the region.

Finally the uncertainties associated with determining the level of storage held in the wetland require discussion. As indicated in Section 5.2.3, the vertical variability observed in subsoil water movement processes was extremely high, resulting in low confidence groundwater level readings. Water moving through preferential flow layers that are intercepted by an observation well would drain into the well to the level of the conducting layer. Groundwater levels would then imply that soil profiles were filled to this level, when in reality underlying layers may have been dry. As total subsoil storage and flow via such pathways (also total water content) is likely to be small compared to that moving via over land flow, through surface peat layers, or through loamy topsoil layers this variability is not considered likely to have introduced large mass balance errors in the modelling work. To improve estimates of wetland filling and draining dynamics in terms of storage held, future studies will need to employ methods such as nested piezometers to evaluate the within wetland variation over annual wetting and drying cycles.

6.3.2.2. Plant functional group modelling

This study has used simple ecological models at functional group level. Thresholds in the models are likely to be indicative of the broad functional group requirements, but require additional work to be of high confidence in an absolute sense. While it is possible (and common) to use a 50% modelled probability as an indication of presence and base conclusions on community composition, the results of the modelling are better interpreted in a more relative sense for at least two reasons. Firstly limited data from only four sites with 120 samples limits the generality. Secondly the models of probability were based on watertable dynamics developed from very short observation periods and were interpolated or extrapolated as necessary to provide measures for all of the observed vegetation communities. These relationships, while physically based, were only subject to qualitative field verification.

The derivation of wetland phreatic surface dynamics from the hydrological models also very likely understates short term dynamics and may understate differences between scenarios that would be observed in reality. Modelled functional group probabilities are likely to be overestimated, in particular for drier scenarios. Future assessments will need to develop more precise methods for monitoring and modelling phreatic surfaces and add sub-models for different life history phases to improve predictive power.

Hydrological processes have characteristic magnitudes, durations and return periods; ecological processes have characteristic probabilities of occurring in response to these. Determining high-confidence thresholds of ecological change is difficult and time-consuming work, requiring detailed hydro-ecological monitoring collected for periods of multiple years. Model refinement requires testing and manipulative experimentation. In reality, the work of quantifying these probabilistic relationships and incorporating this understanding into water allocation planning decisions is at early stages. This general model, like others being developed, will benefit from real world application and as additional data become available to improve confidence in these predictions.

7. FUTURE INVESTIGATIONS

This study has contributed the first quantitative estimates of rainfall-runoff processes in southern Fleurieu Peninsula catchments based on locally collected data. Basic understanding of the manner in which water moves to and through perched wetlands on low order streams has also greatly increased, with contributing processes identified. This work is a basis from which to further improve our understanding of perched wetland ecology and hydrology. Clearly considerable variation in processes between sites is evident and additional investigations to consolidate this new understanding and extend it to landscape scales are now required. Future investigations should focus on both confirming the understanding generated within this work over a longer observation period and extending it to include the remaining knowledge gaps. It is recommended that future work focuses on the following.

Maintain existing monitoring sites to improve the length of the monitoring record and data available for model calibration – Maintain the existing hydrological monitoring network across the three land uses and review the performance of hydrological model parameter sets after additional data are collected. This work could occur with at least an additional two to three years worth of data.

Improve the coverage of wetland monitoring – Install continuous surface and groundwater monitoring in wetlands of higher stream order and undertake soil survey to quantify wetland phreatic surface dynamics and down valley seepage. Implement a soil moisture monitoring network to examine dynamics during the dry season as a variable for input to thresholds models of vegetation tolerance. With this additional understanding, build models of Fleurieu catchments that integrate wetland hydrology, catchment land use and farm dam dynamics and wetland ecology.

Investigate the ecological condition of wetlands along a water resource usage gradient – No information is currently available as to the number of water-stressed wetlands that may be present in the landscape. This study presents a number of hypotheses as to how a water-stressed perched wetland vegetation community would present in terms of the likely plant community but the number of perched wetlands that are actually under stress is unknown. Desktop GIS analysis could be undertaken to determine the likely level of water stress based on the level of both farm dam and forestry development within wetland catchments. This would then provide a means to sub-sample wetlands across the water use gradient (including near natural catchments) to determine development levels where impacts occur.

Map the distribution and prevalence of perennial perched wetlands – The process of calibration (ter Braak 1995), or identifying environmental characteristics from the biological community, offers good promise for mapping wetlands with permanent saturation. Given that perched wetlands in the region number over 500, a remote sensing hyper-spectral mapping approach focussing on plant species requiring saturated conditions is perhaps the most feasible means at regional scale to identify sites where permanent saturation is prominent. One mapping method could involve undertaking a 'wetness inventory' of Fleurieu wetlands using remote survey techniques during late summer–early autumn. Any swamp saturated at surface level, or with standing water is reliant on a local flow system. Once mapped, a subset of wetlands with shallow surface water at this time of year could be investigated through a field program to determine the source of the water and the ecological differences in wetlands as a result.

Improve ecological modelling and wetland watertable and subsoil water movement monitoring and modelling to develop hydrological state and transition models – Key tasks include identifying suitable sites to investigate the movement of water via subsoil cracks (pathway 2). Investigate the possibility of classifying wetland hydro-period based on vegetation community and build models of expected wetland communities under different levels of water availability and which are capable of predicting plant community level response to wetting–drying cycles to assist in planning for extended supra-seasonal drought periods and climate change.

Undertake modelling under a range of wetland: catchment ratio combinations to assess the impacts of worst case and best case situations – This will help to ensure that buffer distances can be determined for all possible variations in the wetland to catchment ratio.

Undertake seed bank germination trials at identified water stressed sites to investigate lost functional groups and to predict recovery potential – Look for opportunities to evaluate improvements in ecological condition at fully forested plantation sites (where considered of conservation value) through the method of larger buffer zones

Determining (if any) the number of wetlands dependent upon outflows from regional fractured rock aquifers and quantifying their hydro-ecological processes – This should occur using hydro-chemical techniques combined with rapid assessment indicators for mapping that will allow managers to easily distinguish these wetlands, which are likely to have different environmental water requirements

Quantifying the hydro-ecology of Permian Sands wetlands – in particular the reliance of such systems on localised runoff and the impacts of watertable surface drawdown from regional scale processes on wetland ecology

APPENDIXES

A. DETAILED SCENARIO MODELLING RESULTS

Table 11. Annual runoff summary statistics – all scenarios

Scenario	Buffer strip (m)	Mean[1] (ML)	Standard deviation	CV[2]	Median[1] (ML)	% of median pasture runoff
12% forestry	10	151.9	78.97	0.52	148.8	92%
30% forestry	10	132.6	71.30	0.54	128.5	79%
50% forestry	10	111.7	65.39	0.59	103.1	64%
70% forestry	10	90.9	62.60	0.69	75.1	46%
100% forestry	10	59.5	64.99	1.09	38.9	24%
12% forestry	20	151.3	78.64	0.52	148.2	91%
30% forestry	20	133.0	71.44	0.54	129.0	79%
50% forestry	20	112.5	65.55	0.58	104.1	64%
70% forestry	20	91.9	62.66	0.68	76.5	47%
100% forestry	20	61.0	64.70	1.06	40.8	25%
12% forestry	50	151.3	78.64	0.52	148.2	91%
30% forestry	50	134.3	71.88	0.54	130.7	80%
50% forestry	50	114.9	66.10	0.58	107.3	66%
70% forestry	50	95.3	62.90	0.66	81.1	50%
100% forestry	50	65.9	63.88	0.97	46.9	29%
12% forestry	100	151.3	78.64	0.52	148.2	91%
30% forestry	100	137.4	73.01	0.53	134.1	83%
50% forestry	100	119.7	67.36	0.56	112.7	69%
70% forestry	100	101.9	63.65	0.62	89.9	55%
100% forestry	100	75.3	62.81	0.83	58.6	36%
Native vegetation	n/a	52.7	33.97	0.65	48.6	30%
Pasture	n/a	164.0	84.51	0.52	162.4	100%

1. As scenarios were run on a hypothetical catchment of area 100 ha, the runoff volume can also be interpreted as a depth in mm.
2. CV = coefficient of variation, the ratio of the standard deviation and the mean

Table 12. Annual runoff volume (ML) for every modelling scenario

Buffer:	n/a	10 metre buffer width (area = 1.4 ha)					20 metre buffer width (area = 2.8 ha)					50 metre buffer width (area = 7.4 ha)					100 metre buffer width (area = 16.3 ha)					Other scenarios	
Year	***Pasture***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***Native vegetation***	***Pine Full***
1973	186	213	187	188	189	190	186	187	188	189	190	186	187	188	189	190	186	186	188	188	190	50	30
1974	279	320	283	285	288	292	280	283	285	288	291	280	282	285	287	291	280	282	285	286	290	87	54
1975	133	153	134	135	136	137	134	134	135	136	137	134	134	135	136	137	134	134	135	135	137	39	32
1976	43	50	43	43	43	43	43	43	43	43	43	43	43	43	43	43	43	43	43	43	43	0	5
1977	64	74	64	64	64	64	64	64	64	64	64	64	64	64	64	64	64	64	64	64	64	10	7
1978	268	307	268	268	268	268	268	268	268	268	268	268	268	268	268	268	268	268	269	268	268	70	47
1979	233	245	178	142	106	51	211	179	143	108	54	211	181	148	114	63	211	187	156	125	79	73	49
1980	132	139	100	79	58	27	119	101	80	59	28	119	102	82	62	33	119	105	87	69	42	41	25
1981	224	234	169	132	95	39	202	169	133	97	42	202	172	137	103	50	202	177	146	114	67	58	37
1982	28	30	21	16	10	3	25	21	16	11	3	25	21	16	11	4	25	22	18	13	7	0	3
1983	288	302	220	176	131	64	261	221	177	133	67	261	224	183	141	78	261	231	193	155	98	101	61
1984	143	149	107	83	59	23	128	108	84	60	25	128	109	87	64	30	128	113	92	72	41	40	21
1985	162	172	127	104	80	45	148	128	105	81	46	148	129	107	85	52	148	133	113	93	63	60	43
1986	173	182	131	103	75	34	156	132	104	77	35	156	133	107	81	42	156	138	114	90	54	40	31
1987	215	228	172	143	115	73	197	172	144	116	75	197	174	147	121	81	197	178	154	130	94	89	71
1988	168	177	129	103	78	39	152	130	104	79	41	152	131	107	83	47	152	135	113	91	59	64	37
1989	234	244	175	136	96	37	210	176	137	98	40	210	178	142	105	49	210	184	151	117	67	62	34
1990	169	177	128	102	75	36	152	129	102	76	37	152	131	106	81	44	152	135	112	89	56	49	34
1991	122	129	94	74	55	26	111	94	75	56	28	111	95	77	59	32	111	98	82	65	41	40	25
1992	389	407	295	233	170	76	351	297	235	173	80	351	301	242	184	95	351	310	257	203	123	120	71
1993	93	100	75	64	52	34	86	76	64	52	35	86	76	65	54	38	86	78	68	58	43	81	33
1994	50	53	39	31	23	11	46	39	31	23	11	46	39	32	24	13	46	40	34	27	17	17	10
1995	158	166	121	97	72	35	143	122	97	73	37	143	123	100	77	43	143	127	106	85	54	47	34
1996	254	268	199	162	125	70	231	199	163	127	72	231	202	168	133	81	231	207	176	145	98	107	67

Buffer:	n/a	10 metre buffer width (area = 1.4 ha)					20 metre buffer width (area = 2.8 ha)					50 metre buffer width (area = 7.4 ha)					100 metre buffer width (area = 16.3 ha)					Other scenarios	
Year	***Pasture***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***12% pine***	***30% pine***	***50% pine***	***70% pine***	***100% pine***	***Native vegetation***	***Pine Full***
1997	89	95	70	58	45	26	81	71	58	46	27	81	71	60	48	30	81	73	63	52	36	35	25
1998	78	82	58	45	32	12	70	59	46	33	13	70	60	47	35	16	70	61	50	39	22	14	11
1999	74	78	56	44	32	14	67	56	45	33	15	67	57	46	35	18	67	59	49	39	24	20	13
2000	252	269	205	174	142	95	233	206	175	144	97	233	207	178	149	105	233	212	186	159	119	118	93
2001	271	288	216	180	143	89	249	217	181	145	91	249	219	185	151	100	249	224	194	162	116	102	86
2002	31	32	22	17	12	3	27	23	17	12	4	27	23	18	13	5	27	24	19	15	8	0	3
2003	183	192	140	111	83	41	165	140	112	85	43	165	142	116	89	49	165	146	122	98	62	54	39
2004	202	213	157	127	97	52	183	157	128	98	54	183	159	131	103	61	183	164	138	112	74	75	50
2005	135	141	102	81	59	26	122	103	82	60	28	122	104	84	64	33	122	107	89	71	43	36	25
2006	56	59	43	35	26	14	51	44	35	27	14	51	44	36	28	16	51	45	38	31	20	16	13
2007	132	138	99	77	54	22	118	99	77	56	23	118	100	80	59	28	118	104	85	66	38	27	20
2008	136	142	101	78	55	21	122	102	79	56	22	122	103	82	60	27	122	107	87	67	37	16	19
2009	221	235	176	146	116	71	203	177	147	118	73	203	178	151	123	80	203	183	158	132	94	91	68

Table 13. Mean monthly streamflow – all scenarios

landuse	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Pasture	0.36	0.01	0.02	0.26	2.07	15.57	39.11	40.30	32.93	21.72	8.12	3.52
Native veg	0.10	0.00	0.00	0.07	0.08	3.86	14.89	12.55	9.64	8.09	2.52	0.86
10B12	0.32	0.00	0.01	0.24	1.78	14.48	36.33	37.28	30.50	20.33	7.48	3.17
10B30	0.26	0.00	0.01	0.20	1.33	12.69	31.90	32.47	26.61	18.07	6.47	2.62
10B50	0.19	0.00	0.00	0.15	0.93	10.64	27.08	27.25	22.42	15.64	5.38	2.05
10B70	0.12	0.00	0.00	0.11	0.63	8.47	22.22	22.04	18.22	13.23	4.34	1.48
10B100	0.03	0.00	0.00	0.04	0.31	5.14	14.61	14.17	11.94	9.66	2.87	0.72
20B12	0.32	0.00	0.01	0.23	1.77	14.42	36.18	37.12	30.37	20.24	7.45	3.15
20B30	0.26	0.00	0.01	0.20	1.34	12.73	32.00	32.57	26.70	18.12	6.49	2.64
20B50	0.19	0.00	0.00	0.15	0.95	10.71	27.26	27.44	22.56	15.73	5.42	2.07
20B70	0.13	0.00	0.00	0.11	0.65	8.58	22.47	22.31	18.43	13.35	4.39	1.51
20B100	0.03	0.00	0.00	0.05	0.31	5.30	15.00	14.54	12.23	9.83	2.93	0.75
50B12	0.32	0.00	0.01	0.23	1.77	14.42	36.18	37.12	30.37	20.24	7.45	3.15
50B30	0.26	0.00	0.01	0.20	1.37	12.84	32.28	32.89	26.95	18.25	6.55	2.67
50B50	0.20	0.00	0.00	0.16	0.99	10.96	27.82	28.05	23.05	16.01	5.54	2.14
50B70	0.14	0.00	0.00	0.12	0.70	8.94	23.26	23.15	19.11	13.74	4.56	1.60
50B100	0.05	0.00	0.00	0.06	0.33	5.83	16.22	15.77	13.20	10.38	3.16	0.86
100B12	0.32	0.00	0.01	0.23	1.77	14.42	36.18	37.12	30.37	20.24	7.45	3.15
100B30	0.27	0.00	0.01	0.21	1.44	13.14	32.99	33.66	27.57	18.62	6.72	2.76
100B50	0.22	0.00	0.00	0.17	1.07	11.44	28.93	29.25	24.02	16.58	5.79	2.27
100B70	0.16	0.00	0.00	0.13	0.79	9.63	24.79	24.79	20.43	14.50	4.88	1.77
100B100	0.07	0.00	0.00	0.08	0.43	6.83	18.51	18.15	15.09	11.44	3.60	1.08

Pine planting fractions and buffer distances presented in the form 'buffer width–B–planting fraction'. Hence a code of 20B30 represents a scenario with a 20 metre buffer and a 30% planting fraction.

B. TANH ANNUAL RUNOFF CURVES FOR SELECTED LAND USES

Rainfall-runoff relationship

Developing an annual rainfall runoff relationship provides a simple means of estimating the volume of annual runoff that can be expected from a catchment for a given amount of rainfall. As rainfall is highly variable through time, there will also be variation in annual runoff generated from the catchment. The runoff coefficient and Tanh function are two commonly used tools in annual rainfall runoff analysis.

The runoff coefficient for the catchment is derived by dividing the average annual runoff by average annual rainfall for the catchment

The Tanh function is a simple rainfall runoff relationship and provides an effective site-based relationship that can be used for infilling annual or monthly runoff values. It is a standard hyperbolic function defined as:

$$Q = (P - L) - F \tanh\left(\frac{(P - L)}{F}\right)$$

where Q = runoff (mm)

P= rainfall (mm)

L = notional loss (mm)

F = notional infiltration (mm)

This equation should be used only where the average storage of soil water is approximately constant i.e. where the notional loss and infiltration are expected to be similar. Also as the method uses average rainfall variable as input data, it cannot give an indication of shorter period variation in runoff generation. The adjustment of the Tanh F value for the ungauged catchments on the basis of slope and land use is subjective, but conforms to general experience. With the lack of calibration data it is not possible to assess the appropriateness for any particular catchment

Determination of F and L

The values of notional loss L and infiltration F are determined by plotting monthly flow sets, seasonal flow sets or annual flows against associated rainfall. A preliminary value of l is chosen from data and F fitted either by trial and error or with a curve fitting technique. Similarly the preliminary estimate of L can be changed to improve the fit.

The runoff coefficient derived with the Tanh curve. The value taken is consistent with the Tanh inputs as referenced to DWLBC GIS coverage. For the whole Fleurieu region, the runoff coefficient ranges from 0.06- 0.17, with higher rainfall zones leading to higher runoff coefficients. The average runoff coefficient taken for research area is 0.12.

Modelled Rainfall - runoff response

Annual modelled runoff under the different land covers and observed rainfall was fitted to a tanh function (Grayson *et al.* 1996). The resulting curve and parameter set from least squares optimisation of model parameters to the data for selected scenarios appears below (Figs 38–42), while parameter values are shown in Table 14. Also shown in the table are predicted depths of runoff for an average annual rainfall typical of the high elevation catchments in the region (947 mm). Runoff volumes predicted by the function at this rainfall depth for the different land uses vary between 33–157 mm resulting in runoff coefficient of 3.5–16.5%.

Table 14. Estimates of predicted depth of runoff for an annual rainfall of 947 mm for selected land uses using the tanh runoff function and modelled data (parameters also shown)

Land cover	Depth of runoff (mm)	Parameter estimates
Pasture	157	L = 86.0; F = 1036.9
12% pine forest, remainder pasture	144	L = 84.4; F = 1096.1
30% pine forest, remainder pasture	126	L = 86.4; F = 1184.4
100% pine forest (20 m buffer)	33	L = 0.0; F = 2870.2
Native vegetation	50	L = 0.0; F = 2320.2

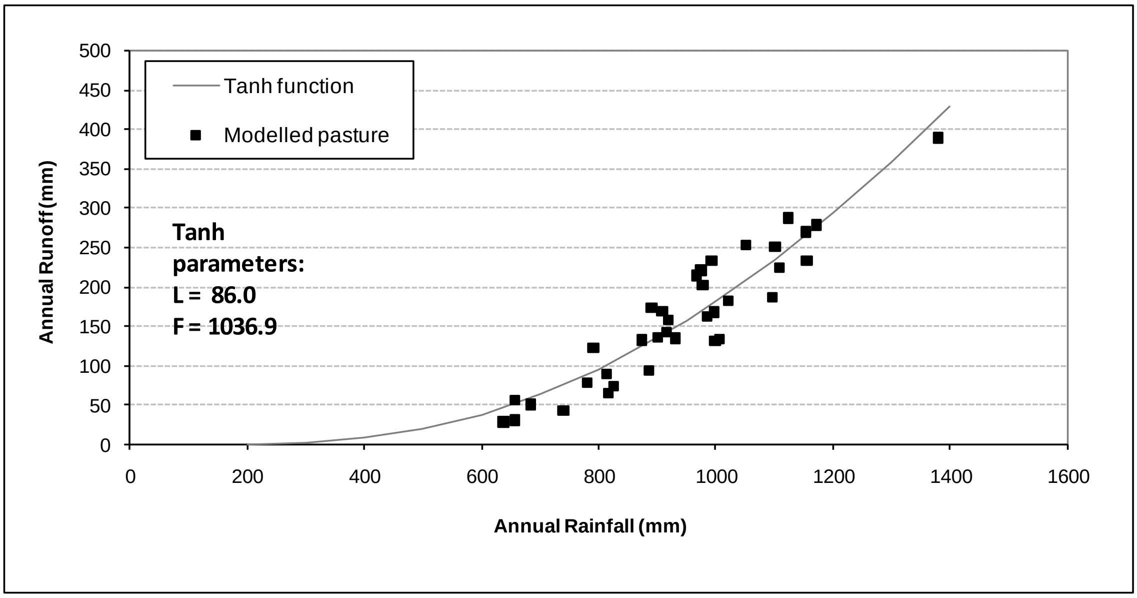


Figure 38. Tanh function fitted to annual runoff data for pasture scenario

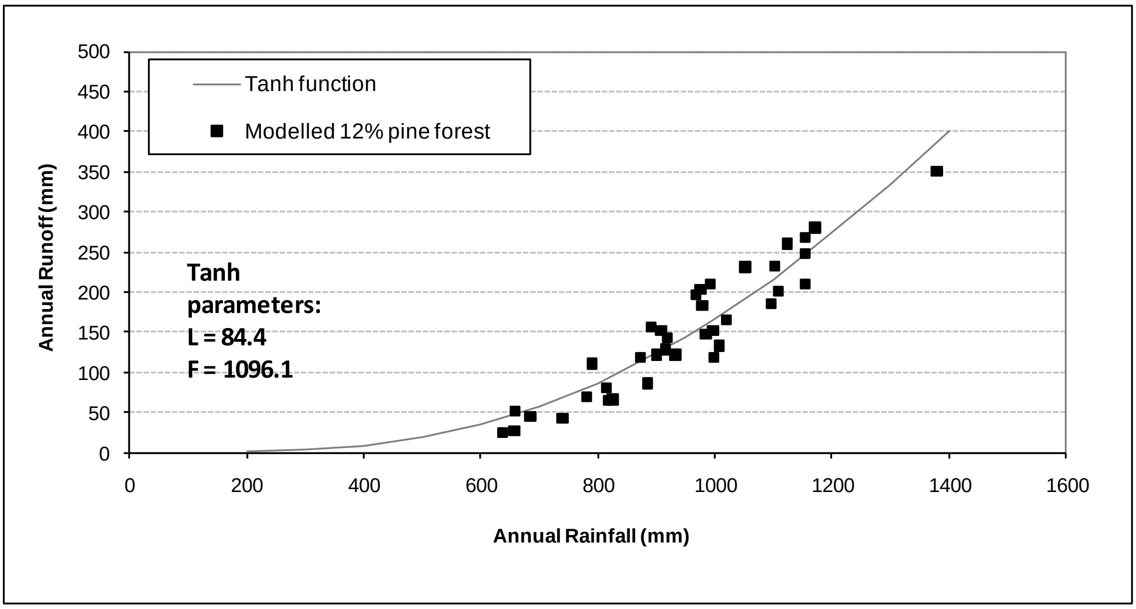


Figure 39. **Tanh function fitted to annual runoff data for 12% pine scenario**

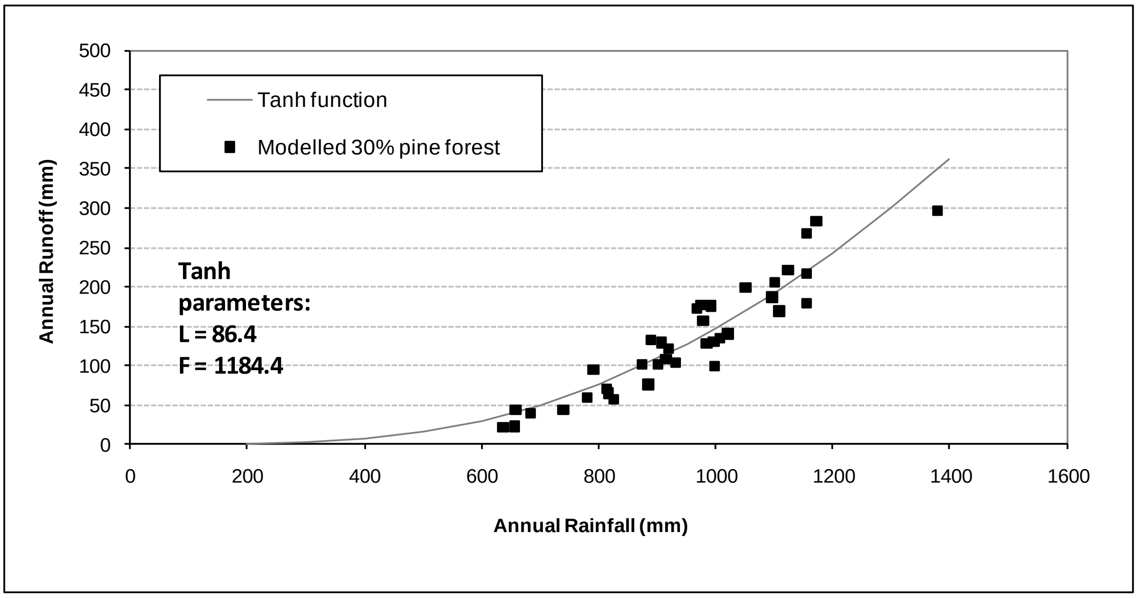


Figure 40. **Tanh function fitted to annual data for 30% Forest scenario (20 m buffer)**

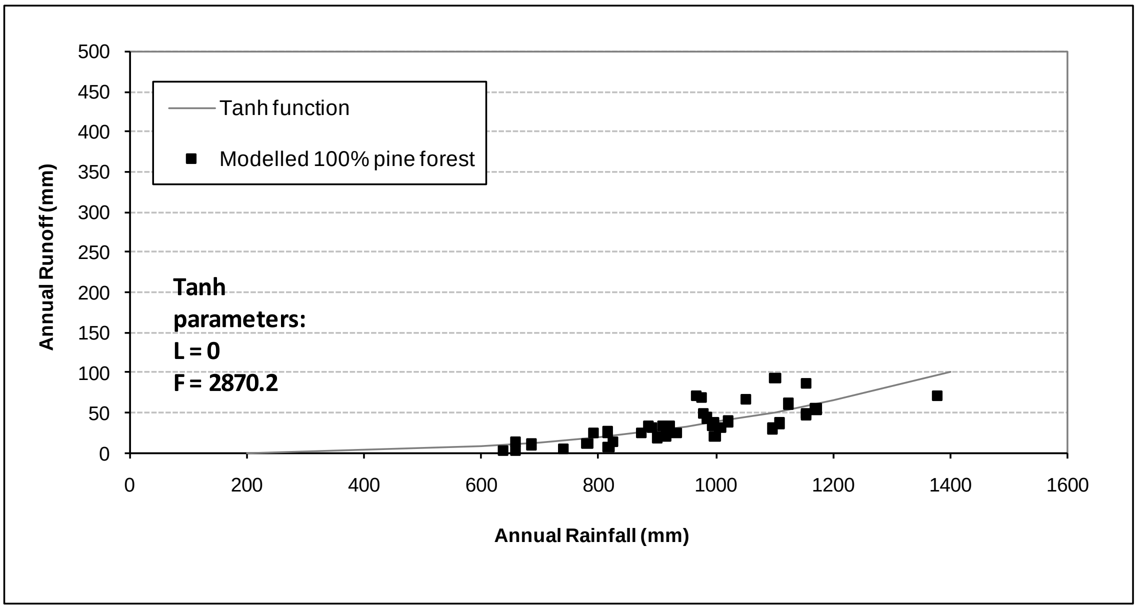


Figure 41. **Tanh function fitted to annual runoff data for 100% pine scenario (20 m buffer)**

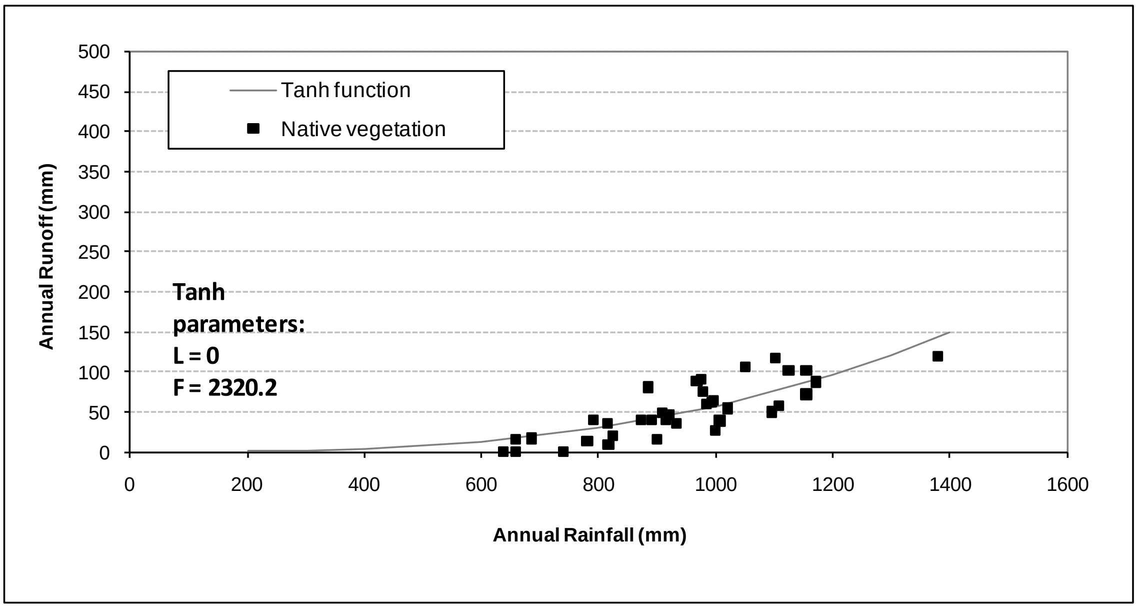


Figure 42. **Tanh function fitted to annual runoff data for native vegetation scenario**

C. BINOMIAL LOGISTIC REGRESSION EQUATIONS

PFG [1]	Linear predictor	95% CI (intercept)[2]	95%CI (β1)[2]	Deviance explained	AIC[3]	CCR[4]	PPP[5]	NPP[6]	Pr(Z > z)[7]
Afte	ln(Pr{Afte}/(1- Pr{Afte}) = 0.030 * d.y.0 - 2.25	-3.37; -1.32	0.019; 0.044	44%	63.6	0.76	0.88	0.69	1.50E-04
Aftw	ln(Pr{Aftw}/(1- Pr{Aftw}) = 0.027 * d.y.1 - 3.37	-5.55; -1.75	0.014; 0.043	33%	67.3	0.76	0.72	0.81	4.83E-04
Afrp	ln(Pr{Afrp}/(1- Pr{Afrp}) = 0.019 * d.y.0 - 1.90	-2.88; -1.08	0.011; 0.029	24%	73.9	0.69	0.82	0.62	2.55E-03
SE	ln(Pr{SE}/(1- Pr{SE}) = 0.031 * d.y.1 -5.31	-9.60; -2.78	0.014; 0.057	27%	64.6	0.91	0.78	0.97	5.64E-06
Tdry	ln(Pr{Tdry}/(1- Pr{Tdry}) = -0.030 * d.y.1 + 3.13	1.50; 5.36	-0.047; -0.018	39%	59.4	0.89	0.95	0.85	3.82E-08
Tdamp	ln(Pr{Tdamp}/(1- Pr{Tdamp}) = 0.028 * d.y.0 - 1.32	-2.24; -0.51	0.016; -0.044	31%	67.6	0.72	0.73	0.72	1.13E-03

1. Plant functional group
2. Confidence interval for the parameter shown
3. Akaike information criterion
4. Correct classification rate
5. Positive predictive power
6. Negative predictive power
7. Probability of observing classification due to chance

D. SOIL DESCRIPTION REPORT EXTRACTS

Soil and landscape description – Foggy Farm

This description is primarily based on detailed soil survey and landscape analysis of the Foggy Farm study catchment, as well as on geological, landscape and soil information contained within DWLBC 2007.

Catchment Description: Remnant plateaux, slopes and wetlands adjacent to and south of Range Road, to the west of Parawa; where Range Road defines the north-south watershed. The study catchment is wholly situated within the Parawa land system (DWLBC 2007).

Geology: The landscape is underlain by metamorphosed siltstones of the Backstairs Passage Formation, which is part of the Kanmantoo Group of rocks that were deposited in shallow seas in the early Cambrian Period (over 500 million years ago). Under soils, these rocks are usually highly weathered, with some deeply weathered areas with kaolin-rich material on the least disturbed and highest-level plateau surfaces.

At Foggy Farm most soils are formed directly in highly weathered siltstone, as there has been considerable natural erosion, owing to relatively steep slopes and relatively high rainfall levels, resulting in reasonably young and shallow silty soils forming in what is, in terms of exposure to soil-forming conditions, comparatively newly exposed rock.

Nonetheless, there is deeply weathered material (kaolinite clay, ironstone gravel and ferricrete) on the high-level plateau areas where the least erosion has occurred, especially on the highest and flattest parts; on other plateaux or summit surfaces, however, there has been significant erosion and ironstone is less evident and deeply weathered layers are often absent. Where ironstone layers do occur, they often occur in the lower part of the soil profile and coincide with the ‘mottled-zone’ where the red mottles have hardened into ironstone.

Most soils on slopes are formed in highly weathered, soft siltstone.

At some wetland sites, there is evidence of deposition that has occurred since European settlement, which includes some charcoal. Surface layers of peat (up to about 50 cm thick, but usually from 10–30 cm) in wetlands are also common; humose-rich loamy to clay loamy topsoils can be very thick. Most wetland soils have formed directly in highly weathered, soft silty rock and are mostly a product of the weathering of that rock. This indicates that removal of material in wetland areas has been more dominant (over recent geological times) than accumulation of materials, pointing to higher rainfall periods than the present and probably periods of meagre vegetation cover (e.g. after fire or, more recently, after heavy grazing).

Topography: What was once a peneplain is now highly dissected plateau, with only meagre parts of the plateau remaining. The catchment comprises remnant plateau areas that are generally level to very gently sloping (1–3% slope), but include some gently sloping areas (3–10%), especially in the form of saddles and upper slopes. Hillslopes between plateaux and wetlands form the main part of the catchment. These range from gentle (3–10%) to moderate (10–32%), with most less than 20%. Lower slopes become noticeably wetter than upper or middle slopes. Wetlands include: gently to very gently inclined creek-like sections, mostly comprising the very lower parts of lower slopes, which are dominated by stringybarks or pink gums; more swamp-like flatter parts of wetlands, with level to very gentle slopes and very poor drainage, which are dominated by tea-tree (especially where peat occurs), sedges, saw-sedges and rushes; while narrow channels and u-shaped depressions occur, usually as higher-level lateral branches to the main wetland area. Elevation: Elevation of the plateau areas varies from about 330 m in the northern parts of the catchment to 290 m in the south. The wetland areas range from 320 m in elevation in the northern upper branches to 265 m at the very bottom of the catchment.

Main features: The Foggy Farm study catchment is a high-elevation area comprising three distinctive elements. Firstly are the plateau areas. Moderately deep to deep soils formed in deeply to highly weathered material are typical; many contain ironstone gravel. These areas are imperfectly drained, with moderately low inherent fertility and high phosphate fixing capacity. Secondly are gentle to moderate hillslopes situated between plateaux and wetlands, which are the most common components of the landscape. Soils on slopes are generally shallow to moderate depth with moderate to moderately low fertility and moderate drainage which grades to imperfect drainage on lower slopes. Plateau and slope soils are acidic to strongly acidic and are susceptible to further acidification. Thirdly is the area of wetland, much of which is swampy. Although wet, there is no evidence of salinity, which indicates reasonable levels of outflow from the catchment. Wetland soils are mostly deep and are commonly neutral to alkaline and sodic in lower parts of the profile (owing to the accumulation of sodium in lower parts of the catchment landscape). Wetlands are covered by regenerated native vegetation and are considered to be important areas of native habitat. Wetland areas are unsuited to agriculture due to excessive wetness, but have been used in the past for grazing, especially in summer when green feed is absent elsewhere.

Main soils

Note that the background and nature of the soil classes J2, K1, N3, etc are fully described in *The Soils of Southern South Australia* (Hall *et al.* 2009).

Plateaux: moderately deep to deep soils with ironstone gravel (or sometimes ferricrete) formed in deeply to highly weathered basement rock:

J2 Ironstone soil.

Main slopes and many plateau areas: shallow to moderately deep soils formed in highly weathered basement rock:

K1-2 Loam over silty clay.

Drainage depressions, some lower slopes and some wetland areas: moderately deep texture contrast soils with strongly mottled and sodic clayey subsoils that grade to mottled colours featuring gley colours and highly weathered basement rock:

F2-N3a Loam over strongly mottled clay.

Wetland areas: deep to moderately deep soils with thick to medium thickness peaty to humose-rich, dark coloured topsoils and grey or gley coloured clayey subsoils that grade to highly weathered, soft basement rock:

N3b Wet grey clay.

Table 15. Foggy Farm soil landscape map unit summary

SLU (soil landscape map units)	Main features
FfZ1 FfZ2 FfZ3	Level to somewhat rounded remnant plateau surfaces, including some very upper slopes, underlain by metasiltstone rock. Underlying rocks are deeply to highly weathered. Slopes are predominantly less than 10%, with most less than 3%. FfZ1 Level to very gently undulating high-level plateau surfaces [sites 01, 71, 83, 90, CH178] FfZ2 Very gently undulating second-level, somewhat rounded summit surfaces [sites: 72] FfZ3 Level to very gently undulating second-level plateau surfaces [sites: 05, 56, 57, 58, 59, 67, 75, 76, 77] Ironstone and/or quartz gravel are common in profiles and on the land surface. Soils are moderately deep to deep and covered by a mulch of pine needles in forest areas. Surface soils mostly consist of loam, with some clay loam and some fine sandy loam. Subsoils are clayey (often light clay which is silty), mostly yellow-brown in colour (with some brown, red-brown and olive-brown layers) and of tight consistency and intermediate structure. Sometimes upper subsoils are sodic/dispersive. Ironstone gravel is common in profiles and occurs in greatest concentrations either in the lower subsoil or in the subsurface layer; gravel in lower layers can be cemented. Soils are strongly acidic to acidic throughout. Main soils are: Loam over silty clay – K1-2 (E–V) (especially on upper slopes and second-level plateau surfaces) Ironstone soil – J2 (E–C) These soils have moderately low natural fertility (partly due to ironstone induced phosphate fixation and partly due to the extent of weathering and leaching). They are usually imperfectly drained. All land is highly exposed. Although productive pasture and forestry soils, potential for more intensive uses is limited.
CNC1 CNC2 CND1 CND2	Gentle to moderate slopes, formed on highly weathered metasiltstones. Most slopes lie between 5% and 20%. CNC1 Moderate erosion potential slopes [sites: 02, 03, 78, 79, 80, 84] CNC2 Moderate erosion potential lower slopes [sites: 60, 61, 68, 69, 81] CND1 Moderately high erosion potential slopes [sites: CH179] CND2 Moderately high erosion potential lower slopes [sites: 73, 91, 92, CH180] Soils are shallow to moderately deep, overlying highly weathered, soft metasiltstone. They usually have loam to fine sandy loam surfaces (with some silty loam) and are covered by a mulch of pine needles in forestry areas. Subsoils are vary from red-brown to olive-brown – mottles and dull and gley colours increase with depth and are also more common on lower slopes and in depressions. Subsoil texture varies from silty clay loam to light clay (with some heavier textures). A pale-coloured subsurface layer occurs in some profiles (especially on lower slopes). Sometimes profiles contain some ironstone gravel in upper layers, while many contain quartz and highly weathered metasiltstone in lower layers. Sodic/dispersive subsoil layers are common, particularly on lower slopes. Soils are generally acidic to strongly acidic, but on lower slopes lower subsoils may be neutral to alkaline. Main soils are: Loam over silty clay – K1-2 (D-V) Loam over strongly mottled clay – F2 (M-L) (on very lower slopes) Soils have moderate to moderately low natural fertility and grade from moderately well drained on upper to middle slopes to imperfectly drained on lower slopes. All soils are prone to acidification. These slopes are generally not suited to cropping because of the combination of high rainfall, erosion potential, cool maturation conditions and, in some areas, exposure. Pasture and pine forestry productivity are potentially high and there is some scope for irrigated horticulture.
CNE	Lateral and upper level drainage depressions and low-lying lower slope areas adjacent to wetlands. These can be narrow landscape features. Slopes mostly lie between 1 and 10%. **CNE** Imperfect to poorly drained low lying areas adjacent to wetlands, including sloping lateral drainage depressions, very lower slopes, flats and upper drainage basins [sites: 04, 61, 66, 85] Soils are mostly moderately deep and similar to other soils in 'CN' soil landscape units, but are generally duller coloured (especially olive-brown) with more mottling and gley colours, while sodic/dispersive subsoil layers are more common. A pale-coloured subsurface layer is common. Surface textures are loam to clay loam. Highly weathered rock and ferruginised gravel commonly occur in profiles. Soils are generally acidic to strongly, but lower subsoils may be neutral to alkaline.

	Main soils: Loam over silty clay – **K1-2** (E) Loam over strongly mottled clay – **F2-N3a** (E) These areas are non-arable and imperfectly to poorly drained.
XnL	Creek-like wetland areas dominated by stringybark or pink gum with sedges, saw-sedges, rushes and bracken fern. **XnL** Poorly to very poorly drained wetlands [sites: 61, 63, 82] Soils are mostly deep with dark coloured and relatively thick loam, fine sandy loam or silty loam surfaces. Subsoils have clay to clay loam textures, while gley or near gley colours are typical (grey, olive, very dark grey, greenish grey and blueish grey); although less dull olive-brown and yellow-brown colours can occur in upper subsoil layers. Subsoils are sodic/dispersive (prismatic structure is likely to be present in almost all cases). Upper soil layers are acidic to strongly acidic, while lower layers are often neutral to alkaline. Of note, is the observation that water fills and moves through the spaces between subsoil aggregates in many subsoil layers – bleached or orange coloured loose sandy to silty grains occur in these saturated spaces, while soil aggregates are darker coloured and less wet (at the time of observation). Main soils: Wet grey soil – **N3b** (V) Loam over strongly mottled clay – **N3a** (C) These areas are non-arable and poorly to very poorly drained.
XnY	Swampy valley flats dominated by sedges, saw-sedges and rushes, also with bracken fern. **XnY** Poorly to very poorly drained valley flats [sites: 65, 66, CH181] Soils are mostly moderately deep, with dark organic rich loam surface soils and pale-coloured subsurface layers. Subsoils are clayey and sodic/dispersive (at least in their upper part) and strongly mottled with gley colours in the lower subsoil (at least). Profiles are generally acidic to strongly acidic in upper layers and neutral to alkaline in lower layers. Owing to the low permeability of the sodic clayey subsoil, water moves laterally through the topsoil and across the soil surface; as a consequence, there is no build up of peat and topsoils are relatively thin owing to natural sheet erosion. Main soil: Loam over strongly mottled clay – **N3a** (D) These areas are non-arable and poorly to very poorly drained. This is an area that water washes over.
XnU	Swamp-like wetland areas dominated by tea-tree, with sedges, saw-sedges and rushes. **XnU** Very poorly drained wetlands [sites: 62, 64, 70, 74, 86, 87, CH182] Soils are mostly deep, with organic-rich, very dark coloured peaty or loam surfaces, which are medium thickness to thick. The subsoil is usually at 40–60 cm depth. Subsoils are predominantly clayey, with olive-brown, olive-grey, greenish grey, dark grey and grey colours, usually with mottles. Layers of yellow-brown to brown with mottles also occur.

Notes:

M = minor extent (<10%).

L = limited extent (10–20%).

C = common extent (20–30%).

E = extensive extent (30–60%).

VE = very extensive extent (60–90%).

D = dominant extent (>90%).

(After French *et al.* 1968).

Definitions of land and soil attribute classifications (e.g. poorly drained), as well as definitions of landscapes and topography (e.g. moderate slopes) can be found in the glossary of Hall *et al.* 2009.

Detailed Soil Profile Descriptions

Note that Australian Soil Classification descriptions are given below in square brackets (see Isbell 2002).

J2 *Ironstone soil* [Ferric Brown Kandosol-Dermosol-Chromosol-Kurosol]

Thin to medium thickness loam to fine sandy loam with fine weak to moderate polyhedral to granular to massive structure grading to yellow-brown kaolinitic clay (mostly light clay) with weak to moderate fine polyhedral to lenticular structure and ironstone gravel (beginning at 20–55 cm), the greatest concentration of which is often in the 'mottled zone' at the base of the profile (which can be cemented, forming ferricrete). Mottling occurs in the subsoil and increases with depth. The substrate consists of a clayey 'mottled-zone' (with red, light grey and olive-brown colours) which is usually silty (beginning at 55–100 cm). The profile is strongly acidic to acidic throughout. [Sites: 01, 57, 58, 71, 75, 77, 90, CH178]

K1-2 *Loam over silty clay* [Mottled Brown-Red Dermosol-Kandosol-Chromosol-Kurosol]

Thin to medium thickness loam, silty loam or fine sandy loam with fine weak to moderate granular to polyhedral to massive structure grading to yellow-brown, olive-brown, olive-yellow, brown or yellow-red silty light clay, silty clay loam, clay loam, light clay, silty light medium clay or sometimes medium clay with fine weak to moderate structure (beginning at 10–55 cm, but mostly at around 30 cm), which in turn grades to highly weathered, mica-rich, yellow-brown, brown-yellow, olive-yellow, olive-brown, olive-grey or red to purple (in high parts of the landscape) siltstone which usually textures as a silty loam or silty clay loam (beginning at 40–80 cm). A 'mottled-zone' of silty clay soft rock underlies some subsoils in higher parts of the landscape. Subsoil and substrate colour are duller (generally more olive) on very lower slopes; while mottling and duller colours (i.e. more olive colours) in the subsoil usually increase with depth. Minor ironstone and/or quartz gravel and/or weathered siltstone fragments can occur in the profile; often the highest concentration of ironstone is in the lower profile; occasionally ferruginised weathered siltstone fragments occur, especially in the subsurface layer. The profile is usually acidic to strongly acidic throughout; although lower layers can be neutral to alkaline on very lower slopes. Upper subsoil can be sodic/dispersive and a pale-coloured subsurface layer may be present, usually where soil is situated on a very lower slope. Water seems not to be able to readily penetrate weathered rock substrate. [Sites: 02, 03, 05, 56, 59, 60, 61, 67, 68, 69, 72, 73, 76, 78, 79, 80, 81, 83, 84, 85, 91, 92, CH179, CH180]

F2-N3a *Loam over strongly mottled clay* [Humose-Natric Sodosolic-Dermosolic Redoxic Hydrosol to Mottled Brown Chromosol-Sodosol-Kurosol-Dermosol]

Medium thickness to thick topsoil, with loam, fine sandy loam, silty loam, loam or clay loam surface soil with fine weak granular to massive structure and usually a pale-coloured subsurface layer, overlying strongly mottled olive-brown to olive-grey to yellow brown silty medium clay with massive to moderate structure (beginning at 20–40 cm), which in turn grades to mottled highly weathered, micaeous siltstone (which textures as a silty clay) with gley colours (grey, blue-grey). The subsoil is usually sodic and dispersive, sometimes highly so. The profile is usually acidic to strongly acidic in upper soil layers and neutral to alkaline in lower layers. Owing to the low permeability of the subsoil, most water moves laterally through topsoil layers and then over the soil surface. [Sites: 04, 60, 61, 65, 66, CH181]

N3b *Wet grey soil* [Peaty-Humose Dermosolic-Kandosolic Hydrosol]

Deep soils with very thick to medium thickness, usually very dark coloured peat or humose-rich loam, clay loam, fine sandy loam or silty loam with fine weak to moderate granular to polyhedral or massive structure grading to grey to olive coloured (very dark grey, dark grey, grey, green-grey, blue-grey, olive-grey, olive-brown) and mottled clay (or sometimes clay loam) with weak to moderate structure (beginning at 25–150 cm), which in turn grades to very highly weathered, micaeous, dark-coloured to olive-brown siltstone at depth which textures as a silty loam to silty light clay. Sodic/dispersive layers occur (which are commonly highly dispersive), often in the lower topsoil as well as the subsoil – it is likely that most subsoils have a primary structure that is prismatic (as viewed in pit CH182). Profiles are often silty and micaeous throughout; and are usually strongly acidic to acidic in surface layers, but can be neutral to alkaline in lower layers. Topsoils can include 10–30 cm of surface peat; humose-rich topsoil can extend to 150 cm, but more usually extends to 20–30 cm – charcoal is a component of some of these layers. Water moves laterally through topsoils layers, but also via inter-aggregate spaces in subsoils (some subsoil layers have moderate-size dark-coloured aggregates surrounded by yellow-brown or white to light grey sandier or siltier material, which is a preferred path for lateral water flow). [Sites: 62, 63, 64, 70, 74, 82, 86, 87, CH182]

Notes: Soil profile description and terminology are explained in National Committee on Soil and Terrain 2009 and Isbell 2002.

Soil Characterisation Sites: Layer thickness and texture

These excavated sites were fully described (according to National Committee on Soil and Terrain 2009) and comprehensive chemical analyses were conducted on samples from all described soil horizons (according to Rayment and Lyons 2010). Only very brief summaries of morphological descriptions are presented here.

Note the predominance of silty textures ("z" prefix), which indicates a young soil forming or derived from weathered rock, which still has significant mica that hasn't yet broken down to clay. Also note the predominance of soils with sodic (dispersive) layers, which is particularly important for clay loamy (>20% clay) and clayey (>35% clay) layers because of the adverse effect on soil structure and drainage. In some subsoil and other layers of wetland soils there is evidence of preferential throughflow around the mostly prismatic peds created by sodic conditions (e.g. see CH177 and CH182 descriptions below).

See National Committee on Soil and Terrain (2009) for explanations on horizon and texture designations. Texture designation codes used below are: s = sand; fsl = fine sandy loam; l = loam; fscl = fine sandy clay loam; scl = sandy clay loam; cl = clay loam; lc = light clay; lmc = light medium clay; fslmc = fine sandy light medium clay; mc = medium clay; fsmc = fine sandy medium clay; mhc = medium heavy clay; z = silty; + = heavy; - = light; A = sapric (organic); lp = loamy peat.

Wither Swamp

CH174 – remnant plateau surface

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	12	l	20	significant ironstone gravel
2	BA	28	fscl	28	sodic / abundant ironstone gravel
3	Bt	51	lc	38	
4	CBcm	74	-	-	thick ironstone
5	CBr	117	cl+	33	
6	Cr	160	zcl+	33	weathered rock

CH175 – mid to lower slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	10	l	20	
2	B/A	22	lc	38	sodic
3	Bt1	39	zmc	>45	
4	Bt2	59	zlmc	43	
5	BC	77	zlc	38	
6	CBr	100	zcl	30	
7	Cr	150	zcl	30	weathered rock

Wither Swamp (continued)

CH176 – lower slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A1	15	l (A)	20	highly organic
2	A2e	33	fsl	15	bleached / wet at time of sampling: acts as “non-Newtonian liquid”
3	Bt	55	lc	38	sodic
4	CBr	90	cl	30	
5	Cr	140	zcl	30	weathered rock

CH177 – wetland

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	20	l	20	wet at time of sampling
2	A2e	41	scl-	23	wet at time of sampling / bleached material around peds
3	Bt	68	fslmc	43	highly sodic/ bleached material around peds
4	CBr	95	zlc	38	sodic

Springs Rd

CH183 – remnant plateau surface

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	9	l+	23	significant ironstone gravel
2	AB	22	cl-	28	significant ironstone gravel
3	BA	50	lc	38	sodic
4	Bt	78	lmc	43	significant ironstone gravel
5	CB	105	lc	38	significant ironstone gravel
6	Ccm	145	lc+	40	abundant ironstone

CH184 – upper slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	1	fsl	15	
2	BA1	13	fscl-	23	sodic
3	BA2	24	lc	38	sodic
4	Bt	50	lmc	43	sodic
5	CBr	75	fszlmc	43	
6	Cr1	115	fszcl	30	weathered rock
7	Cr2	170	zcl	30	weathered rock

Springs Road (continued)

CH185 – lower slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	9	fsl	15	
2	AB1	26	fscl-	23	significant ironstone gravel
3	AB2	43	fscl	30	significant ironstone gravel
4	Bt	63	lc	38	sodic
5	BCr	82	zlmc	43	
6	CBr	127	zlmc	43	
7	Cr	160	zlc	38	weathered rock

CH186 – wetland

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A11	15	l	20	highly organic
2	A12	33	cl-	28	wet at time of sampling / highly organic
3	AB	50	lmc	43	sodic
4	BA	95	mc	>45	sodic
5	BC	120	mhc	>50	sodic
6	C	145	lc	38	sodic

Foggy Farm

CH178 – remnant plateau surface

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A	12	l	20	significant ironstone
2	BA	28	cl-	28	sodic / significant ironstone
3	Bt	42	cl+	33	sodic
4	BC	80	zcl+	33	
5	CBcm	120	zlc	38	abundant ironstone
6	CBc	155	zcl	30	significant ironstone

CH179 – upper slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A1	10	l	20	
2	AB	32	cl	30	
3	Bt	60	mc	>45	sodic
4	BC	90	zcl	30	
5	B/Cr	115	zcl	30	
6	Cr	165	zl	20	weathered rock

Foggy Farm (continued)

CH180 – lower slope

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A1	8	l	20	
2	BA	20	zcl	30	
3	Bt	50	zlc	38	sodic
4	BC	100	zlc	38	
5	Cr	165	zcl	30	weathered rock

CH181 – wetland

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	A1	12	l	20	wet at time of sampling
2	A2e	25	fscl-	23	wet at time of sampling: acts as "non-Newtonian liquid"
3	BA	46	fslc	38	highly sodic
4	Bt	78	fslmc	43	sodic
5	BCr	120	zlmc	43	

CH182 – wetland. Note: no layer is saturated; water flows around peds

Layer	Horizon	Depth (cm)	Texture	Approx Clay (%)	Notes
1	P2	12	lp		peat
2	A11	28	l+ (A)		highly organic
3	A12	39	cl (A)		highly organic
4	BA	60	lc		sodic / main water flow around prismatic peds
5	B	80	lmc		sodic / main water flow around prismatic peds
6	Bg	100	fsmc		sodic

E. HYDROLOGICAL ANALYSIS FOR THE SOUTHERN FLEURIEU

The data analysis was done to find catchment characteristics. The catchments further referred as 'research catchment' are a representation of the catchments under study (having perched hydrogeological system and catchments having Permian sand systems. All research catchments are shown in Figure 43 below. At the later stage of project (the modelling), the study was limited only to three research area with perched swamps—the Foggy Farm, the Wither Swamp and the Springs Road Native Forest Reserve. Other perched swamps Ballaparudda, Boat Harbor and Deep Creek.

"Research Catchment" or "catchment" in this context refers to the area representing the study area in general.

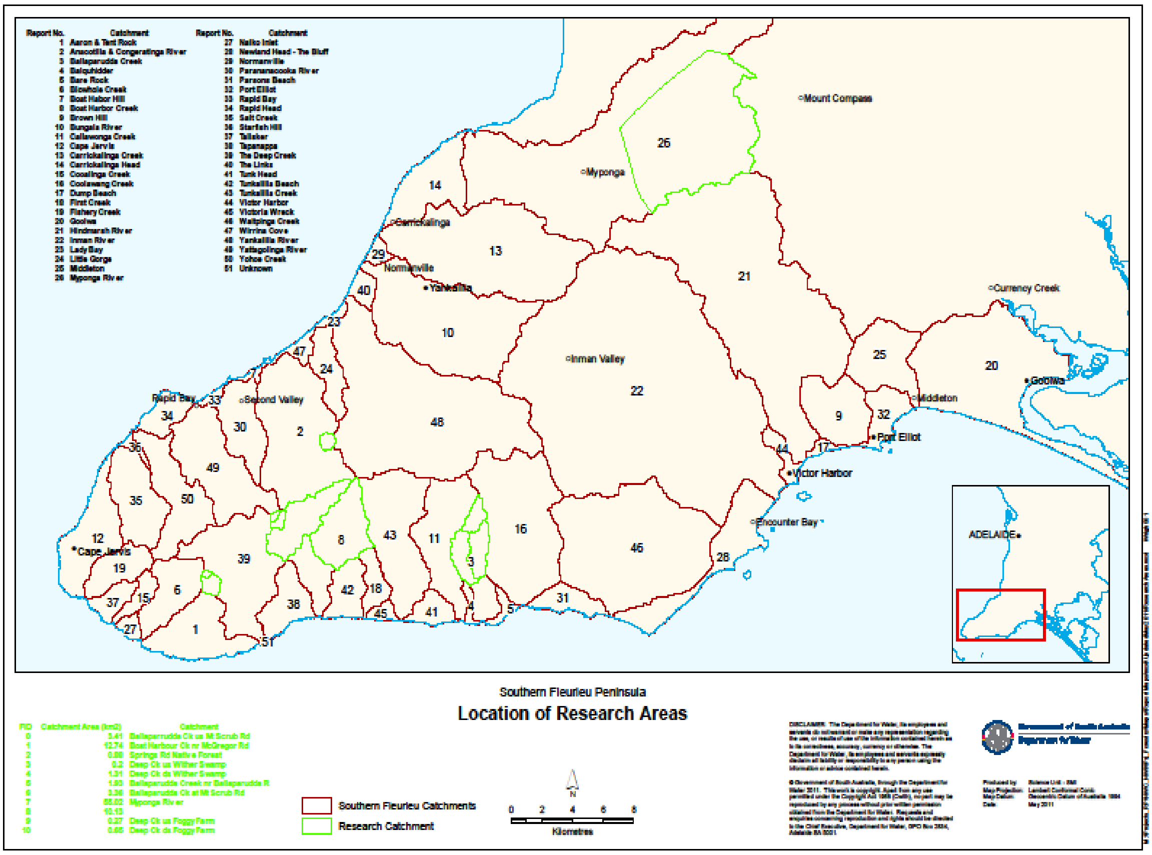


Figure 43. **Location of initial research catchment (areas)**

Rainfall

Data availability and processing

Daily rainfall data in South Australia are being collected by Bureau of Meteorology (BoM) and DWLBC. Figure 2 shows the locations of rain stations in the Fleurieu Peninsula and are listed along with their record period in Table 16. Since the distribution of rainfall stations is more concentrated in the upper part of Fleurieu area and sparse in the lower area, data analysis will just be an indication of the trend in the lower area of Fleurieu Peninsula area.

Table 16. Rainfall stations in Fleurieu Peninsula

	Station no.	Locations	Start date	End date
1	M023708	Second Valley (Spring Grove)	31/12/1877	31/12/1983
2	M023761	Parawa (Sharon)	28/02/1947	31/03/2008
3	M023743	Victor Harbor (Rivington Grange)	31/07/1910	29/02/2008
4	M023751	Victor Harbor	31/12/1883	31/03/2003
5	M023823	Hindmarsh Valley (Fernbrook)	31/12/1964	29/02/2008
6	M023824	Hindmarsh Valley (Springmount)	31/08/1952	31/01/2002
7	M023723	Yankalilla (Inman Valley)	31/12/1932	31/03/2008
8	M023754	Yankalilla	29/02/1892	31/03/2008
9	M023738	Myponga	31/12/1913	31/03/2008
10	M023783	Myponga Reservoir	28/12/1963	29/02/2008
11	M023851	Myponga Hill (Lovely Valley)	22/10/1992	14/02/2007
12	M023850	Hindmarsh Tiers	13/01/1993	14/02/2007
13	M023814	Cape Jervis (Hunt)	31/01/1969	31/12/1985
14	M023815	Rapid Bay BHP	30/04/1970	31/07/1979
15	M023744	Second Valley (Poolamacca)	31/12/1883	31/01/2008
16	M023816	Second Valley Forest	31/12/1969	31/03/2005
17	M023875	Parawa (Second Valley Forest AWS)	31/12/1993	31/03/2008
18	M023762	Willow Creek	31/12/1953	31/12/1987
19	M023811	Second Valley	31/01/1969	30/06/1997
20	M023741	Normanville	31/12/1883	31/01/2008
21	M023782	Yankalilla (Woodvale)	28/02/1870	31/12/1927
22	M023749	Torrens Vale	31/07/1901	30/11/1948
23	M023700	Aldinga Post Office	31/07/1901	30/11/1948

Missing data

The majority of data sources contain missing segments. Missing data in daily rainfall records may occur in two forms: when a value has not been recorded in particular day(s) but a cumulative value has been recorded on a subsequent day; or where the data is missing due to a recording error.

The first type is referred to as accumulated records and these may be disaggregated over the total number of missing days. This disaggregation of data is based on the method described in Porter and Ladson (1993). This assumes that the influence of the rainfall at nearby stations to the stations where data is to be disaggregated is inversely proportional to the distance between the stations. Using nearby stations reduces the uncertainty from using data from single station.

Stations with missing data, due to recording errors, were in filled using data from a nearby station. The nearby station was chosen as the one with the highest correlation between daily values that had data concurrent with the missing period. To infill the missing period, the daily rainfall value at the nearby station was then adjusted by the ratio of concurrent mean annual rainfalls at two stations.

Data consistency

To identify the occurrence, magnitude and nature of trends within long time series records, the double mass curve technique is often used. It is constructed by plotting the accumulated values of two time series against each other. A break in slope or a gradual change in curvature will reveal a change in the constant proportionality between the two sets of data. This indicates the presence of trend such as in measured rainfall due to localised station conditions. The method is often used to establish the presence of such changes within rainfall records and adjustments can subsequently been made to the affected datasets to ensure consistency of record.

In this study, the consistency of each rainfall was confirmed by constructing a double mass curve using an average of monthly rainfall from eight to ten neighbouring stations. Using an average of a number of records reduces inconsistencies that may be present in any one record.

Data Analysis

Analysis of the rainfall data at each station was undertaken at monthly and annual time scales. The rainfall regime and its variability at each time scale affect the runoff and surface water availability differently. As such, all the stations are not used partly because some of them do not have recent data and partly because the stations are relatively far from the study area such that the effects are likely to be insignificant.

To obtain a single record more representative of condition across the study area, 3 datasets were combined. The rainfall stations used for the analysis are M023875, M023816 and M023761. The daily rainfall correlation of M023761 with M023875 is 0.78 and with M023816 is 0.73. Since M023761 has the longest record (1907-2007) with 1947-2007 recorded data and extended back to 1907, this station is used for further analysis.

The Catchment rainfall is calculated using a rainfall factor derived from the relative position and values of isohyets. The rainfall factor is defined as the ratio of the isohyets passing through the study catchment to the isohyets passing through the rainfall station. The rainfall factor is calculated as 0.902.

Annual rainfall

Table 17 shows nearby stations and annual statistics for the study area. The standard deviation provides an indication as to the variability of annual rainfall; the lower the standard deviation, the less is the variability of annual totals around the mean. The representative derived annual rainfall record is shown below for the period 1907– 2007 (Fig. 44) .

Table 17. Annual Statistics of rainfall stations used for analysis

Site No	Station no:	Mean	Median	Std. Deviation
1	M023816 (1970–2005)	937.1	916.2	171.9
2	M023875 (1994–2007)	840.8	856.1	134.2
3	M023761 (1907–2007)*	933.3	925	179.8
	Catchment rainfall	858.7	851	165.4

*recorded data 1947–2007 and data extended back to 1907

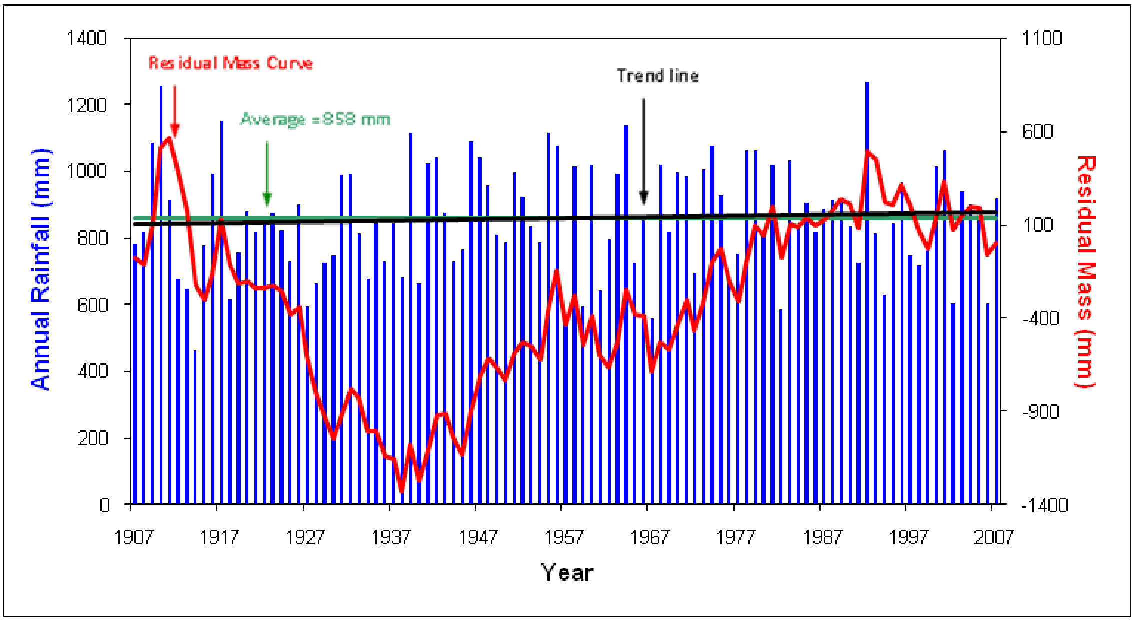


Figure 44. Generalised annual rainfall totals and variability for the study area

Monthly rainfall

Representative mean monthly rainfall data for the study region is shown in Figure 45. Most of the rainfall occurs from April to October with the highest rainfall value in the month of July. Comparative observation of research catchment rainfall with one of the more established rainfall station is shown in Figure 46. The pattern of winter spring dominance is consistent with rainfall station for Myponga (M023738) over the period of 1914–2007. The mean annual rainfall for Myponga is 771 mm and that of catchment is 858.7 mm. The monthly rainfalls for each station are shown below (Figs 45–49).

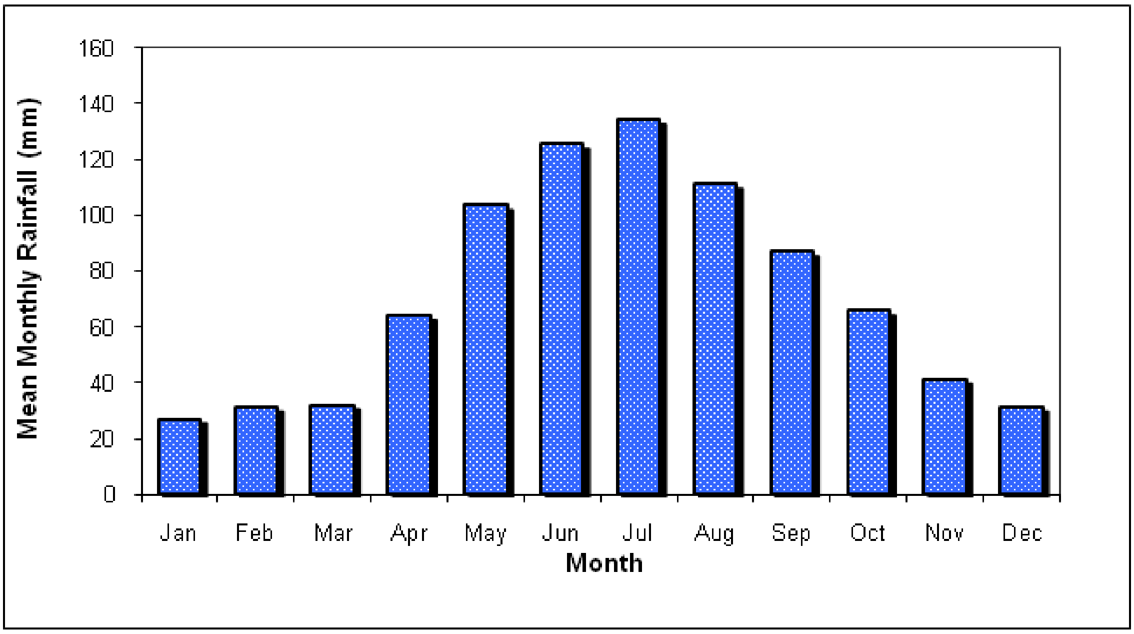


Figure 45. Mean monthly rainfall for the research catchment

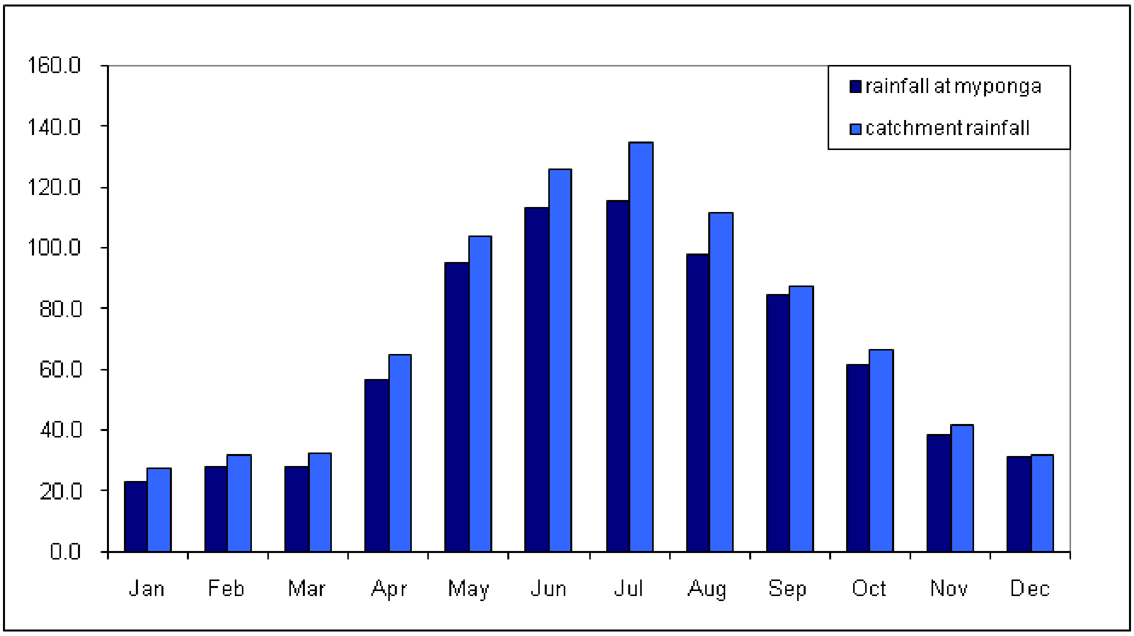


Figure 46. **Monthly rainfall variability for the research catchment and station M023738**

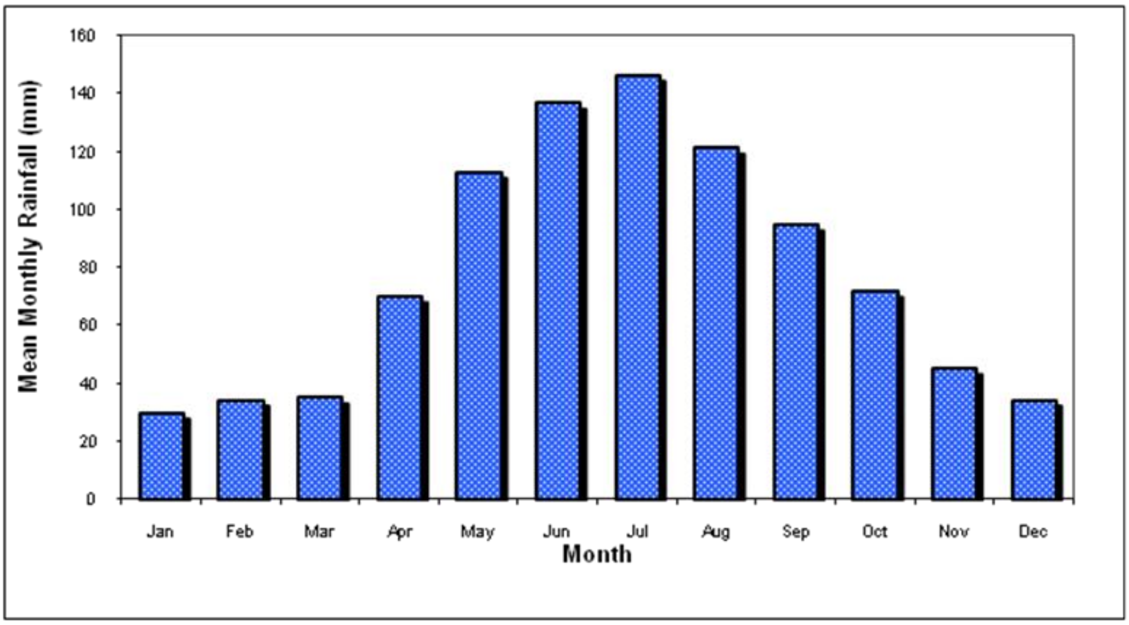


Figure 47. **Mean monthly rainfall for station M023761**

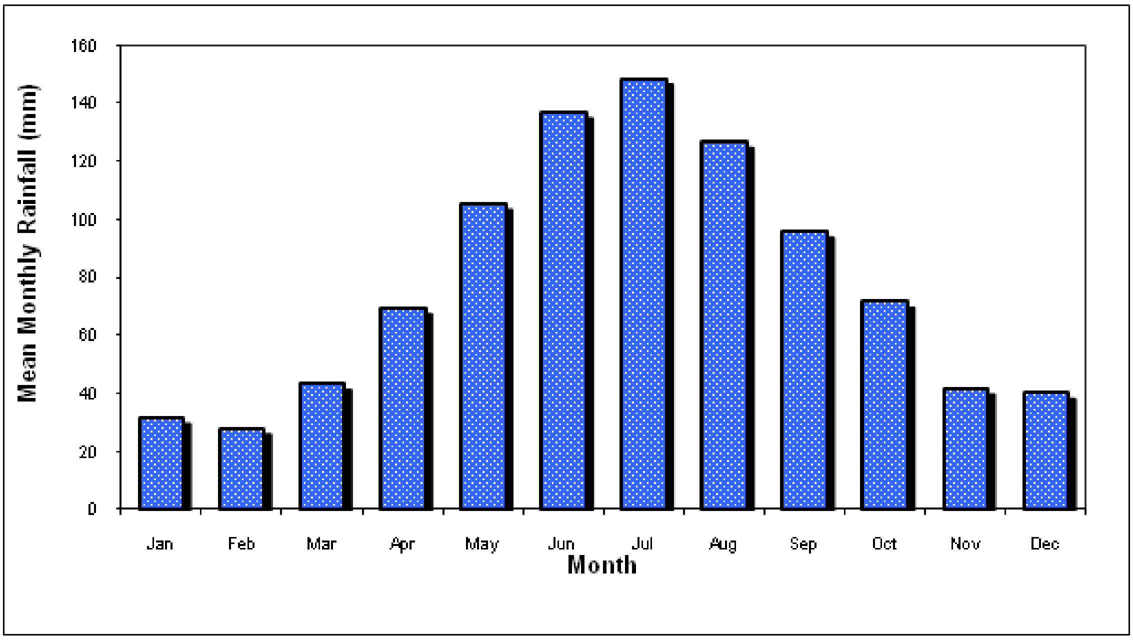


Figure 48. **Mean Monthly rainfall for station M023816**

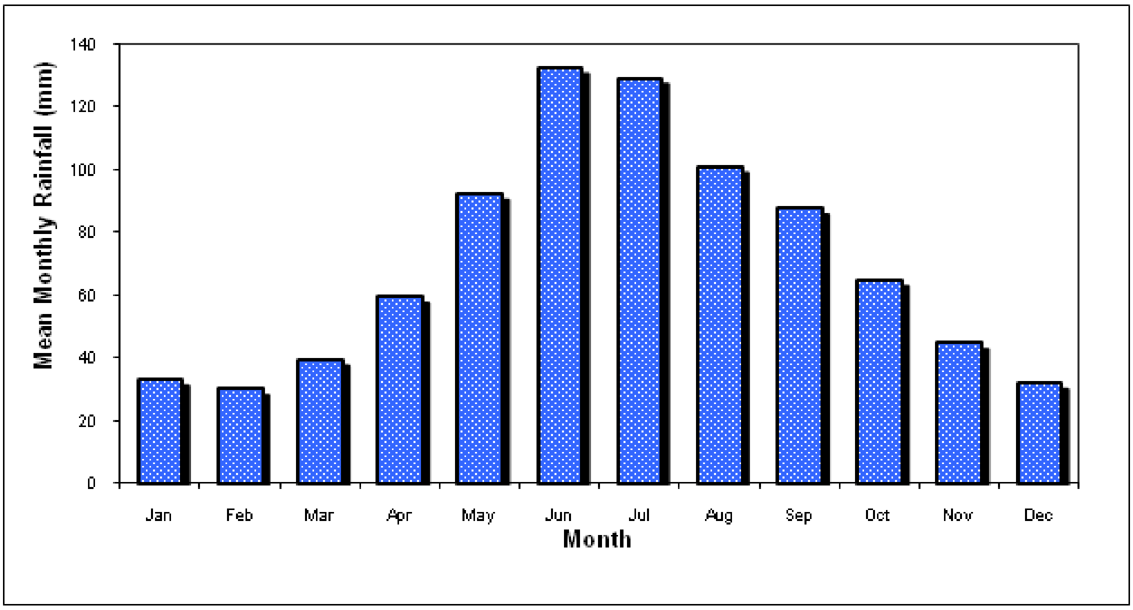


Figure 49. **Mean monthly rainfall for station M023875**

Evaporation

Evaporation is simply the conversion of water from liquid to vapour. Solar radiation, air temperature, air humidity and wind speed all affect the rate of evaporation. Accurate estimates are essential for hydrologic water balance calculations because evaporation influences the amount of rainfall that is intercepted by vegetation and absorbed by soil before surface runoff will occur. It also significantly reduces the volume of water stored in large water bodies, particularly during summer.

Data availability and processing

The availability of daily evaporation data is limited, particularly in comparison to rainfall data. The mean monthly values for evaporation are generally used in the lack of daily data, but the limitation with this use is the large day-to-day variance in evaporation that often occurs during spring and autumn. These discrepancies may cause serious errors in the estimates of evaporation and hence the resulting water balance.

There are two sources of evaporation data, namely measured pan evaporation and evaporation calculated using empirical equations. The method chosen usually depends on the type of surface from which evaporation is occurring because the factors affecting evaporation differs between a dam or a reservoir surface compared to a soil or plant surface.

Pan evaporation is based on evaporation from an open water surface and provides an index of the integrated effects on evaporation from solar radiation, air temperature, humidity and wind speed.

Empirical equations used to calculate evaporation incorporates estimates of the main factors that affect evaporation. Datasets of solar radiation and temperature in particular are often more readily available, have longer data records and less missing data than pan evaporation records. One approach that is commonly used is the Priestley-Taylor method. This has been used for numerous catchment-modelling studies in Australia.

The Priestley-Taylor method uses solar radiation data but because solar radiation is not widely monitored in Australia, it often has to be estimated, spatially interpolated or extrapolated using relationship between solar radiation and sunshine hours and temperature.

Data Analysis

The only evaporation data available in the Fleurieu area is for station no: M023783 (1999–2006). Since only limited data for meteorological parameters are available, the data from station M023783 is used in the following analysis instead of using empirical equations.

The monthly evaporation chart (Fig. 50) shows that the minimum evaporation occurs in June and July. The month of January has the highest evaporation of about 200 mm. In general, the high rainfall months have lower pan evaporation.

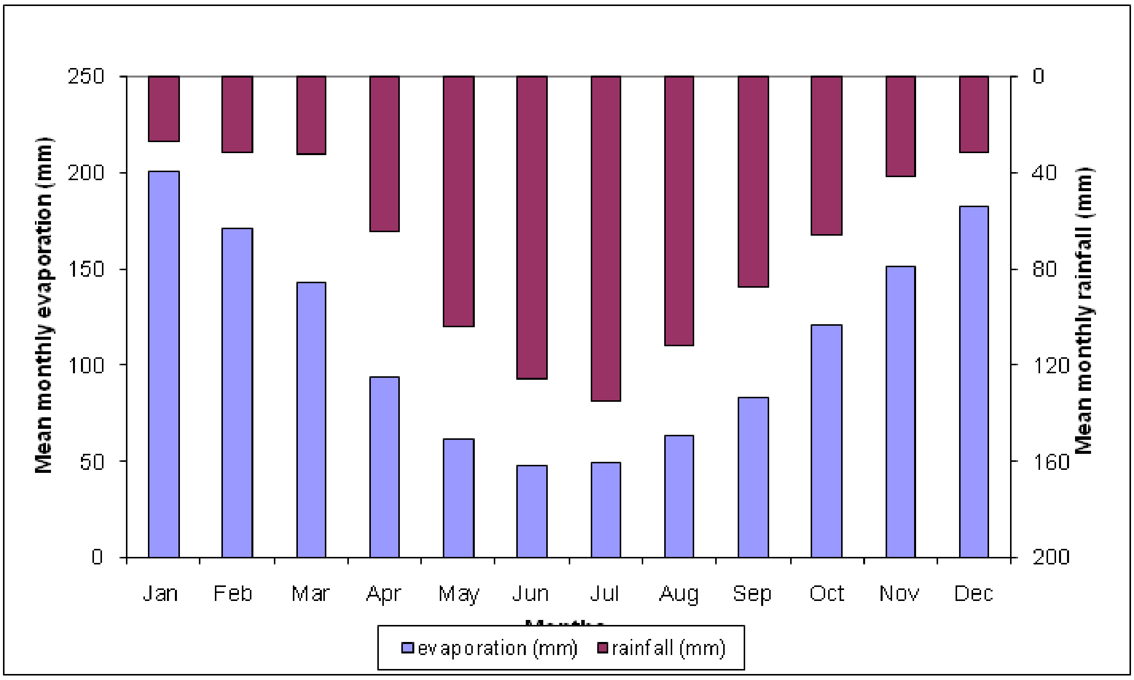


Figure 50. **Mean monthly evaporation and catchment rainfall**

Stream flow data

Gauging stations in the Fleurieu Peninsula are very limited in number and stations with long term data are concentrated in the upper part of the area. While there are newly installed gauging stations in the lower region, none of these stations have a record exceeding two years. Correlation techniques can be used for the extrapolation of data for required stations where suitable sites can be identified and a correlation for the shared period of record established.

The stations that are relevant to research catchments areas are A5011011, A5011010, A5011007, A5011008 and A5011009 (shown in Figure 10). As none of these stations have data of more than one full year, the data is insufficient for analysis. Linear regression analysis shows that correlation between these stations with the nearby stations A5010503, A5011006, A5010500, A5020502 is poor. So, it is not currently possible to derive long-term data for relevant stations. The highest correlation shown by relevant stations is the correlation of A5011007 with A5011006, which had value of R^2 as 0.6055. Most of the other correlations have a value less than 0.5.

While the data analysis below was undertaken in 2008, flow duration curves for all available flow data at the time of report preparation (2006–10) is presented for the three stations as time weighted flow duration curves (Figs 51–53).

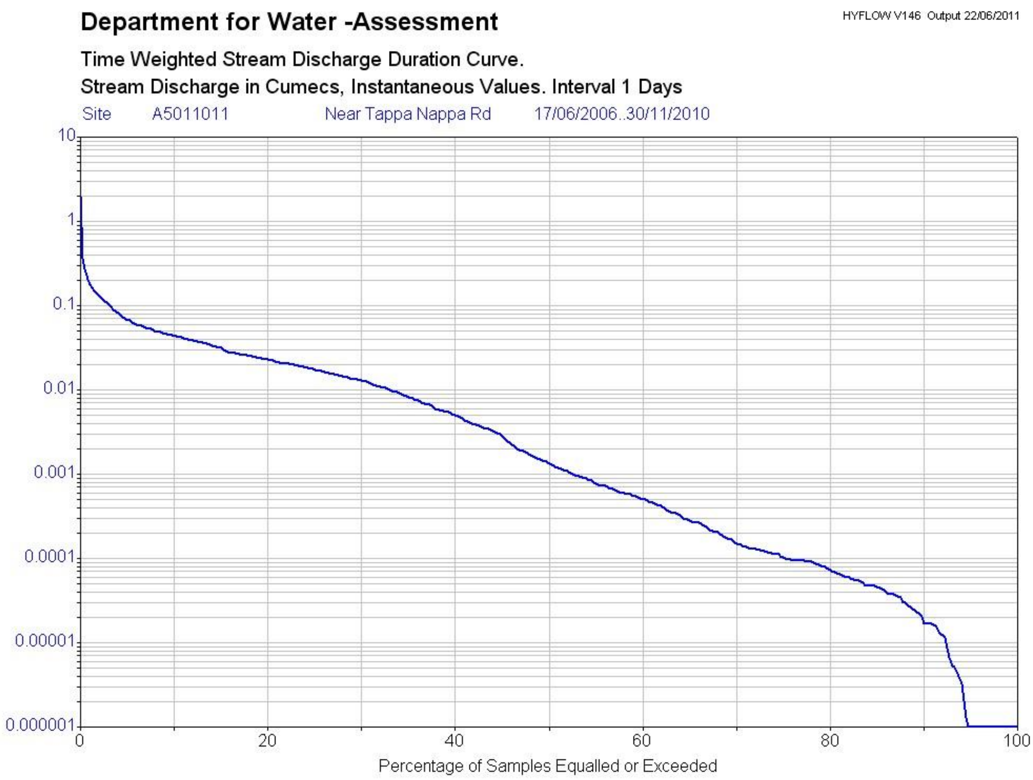


Figure 51. Flow duration curve – Upper Deep Creek catchment (station A5011011)

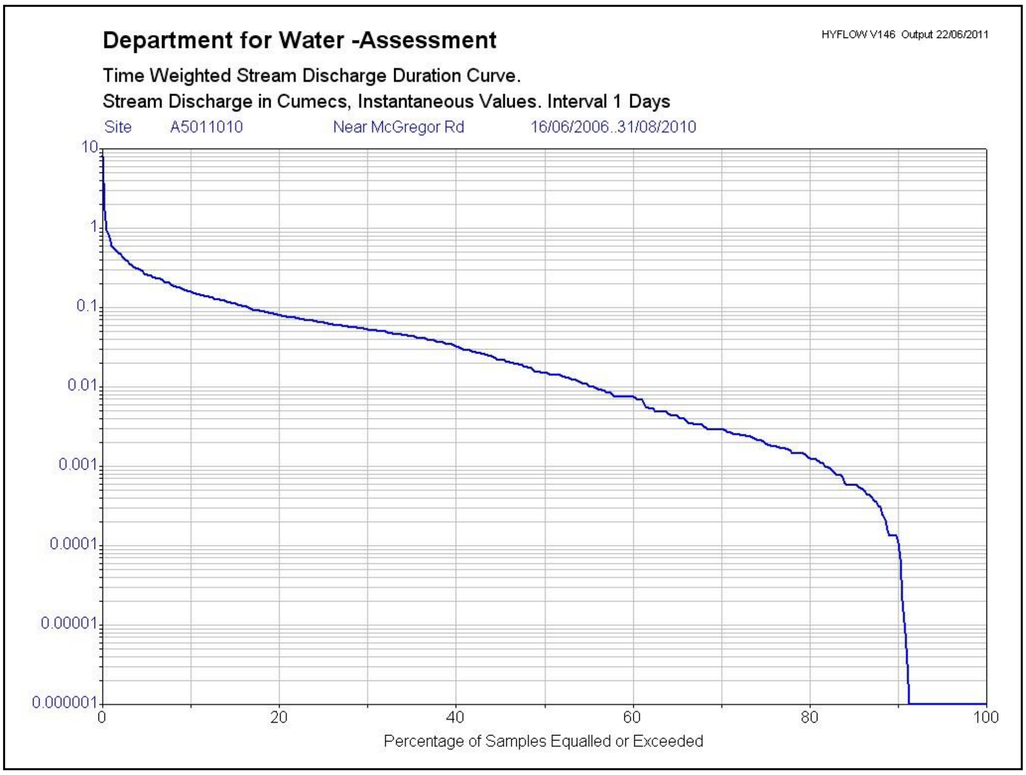


Figure 52. Flow duration curve – Boat Harbor Creek (station A5011010)

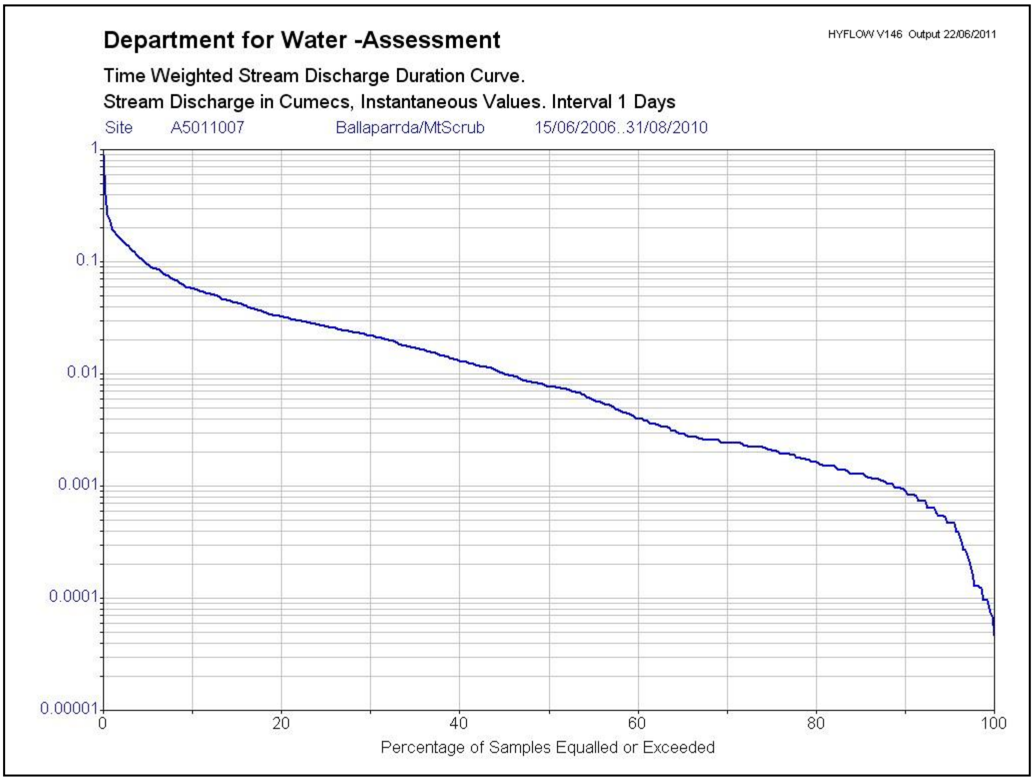


Figure 53. Flow duration curve – Ballaparudda Creek (station A5011007)

Data Analysis

The data availability of the relevant gauging stations is for only a full year (2007). Therefore it is not possible to perform any statistical analysis for annual stream flow. Figure 54 shows the monthly flow compared to rainfall in the catchment for station A5011007 and same has been other station also. It is observed that 90 percent of flow during 2007 occurred between May and November.

However, this is just an indication for the catchment as there is only one full year (2007) of data available for stream discharge. Mean monthly rainfall is taken from 100-year period (1907-2007). The pattern of the stream discharge is justified by the discharge comparison of the station with one of the stations having long-term record (station A5020502, Myponga 1980–2007) in Figure 55.

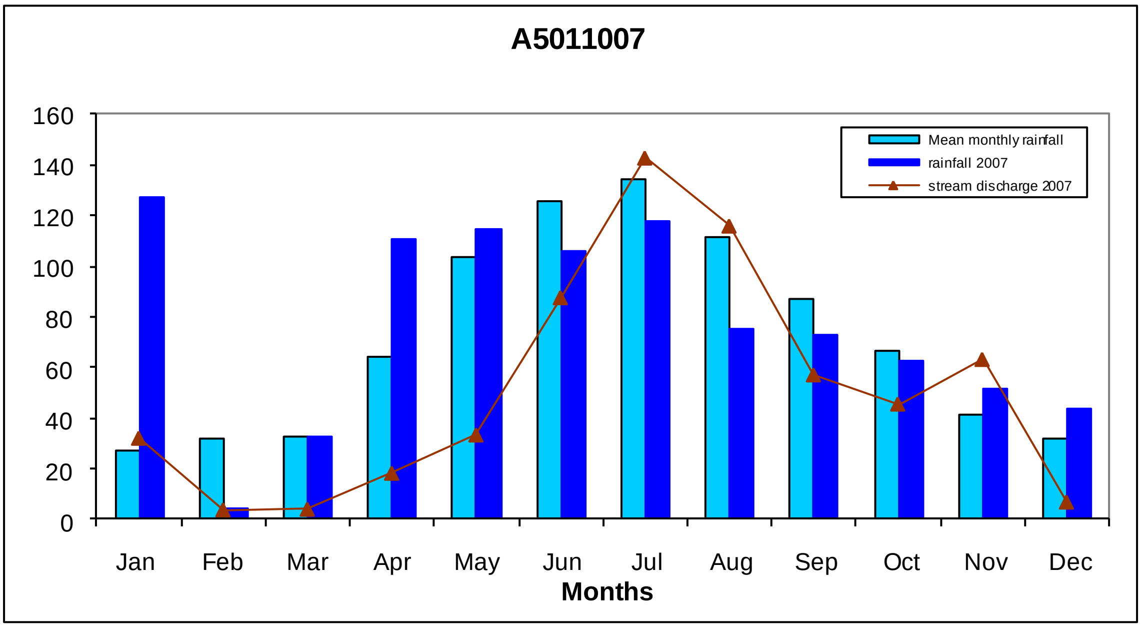


Figure 54. **Mean monthly rainfall and monthly stream flow of station A5011007**

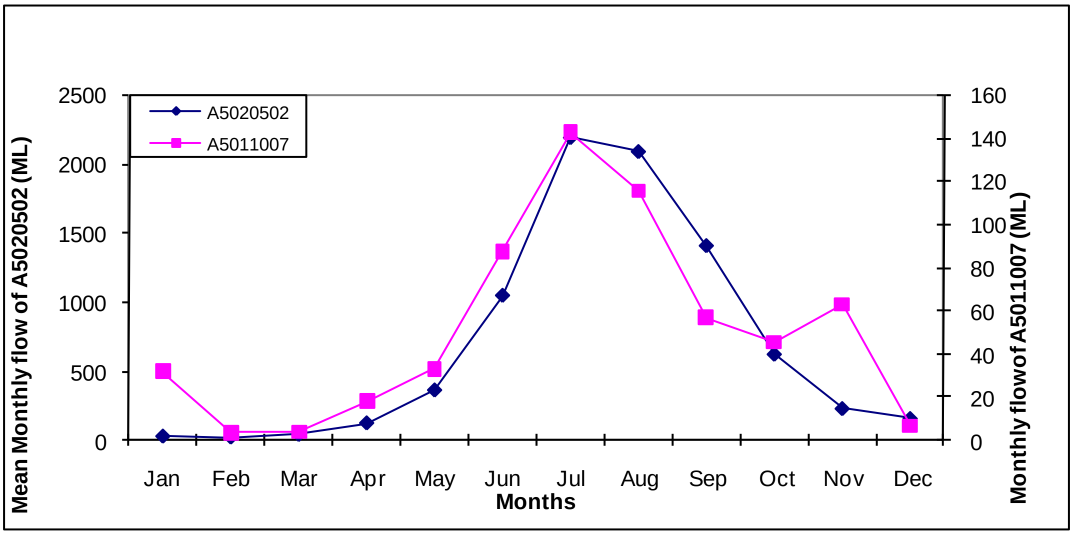


Figure 55. **Comparison of stream flow pattern of station A5011007 with station A5020502**

Daily flow statistics

Daily flow statistics of the available data were used to develop a flow frequency curve. This shows the percentage of time the specified flows were equalled or exceeded during the period of record and provide insight into the volume of water that may occur during a year. Due to lack of data, this analysis is also made using the latest one-year data 2007. Figure 56 shows the frequency curve for gauging station A5011007 and the corresponding characteristics are shown in Table 18. It can be concluded from the figure that there is stream discharge for the whole period of the year and that is >0.1 ML for 96% of time, however, it is not the case for the other stations. The flow frequency curve for each station has been performed.

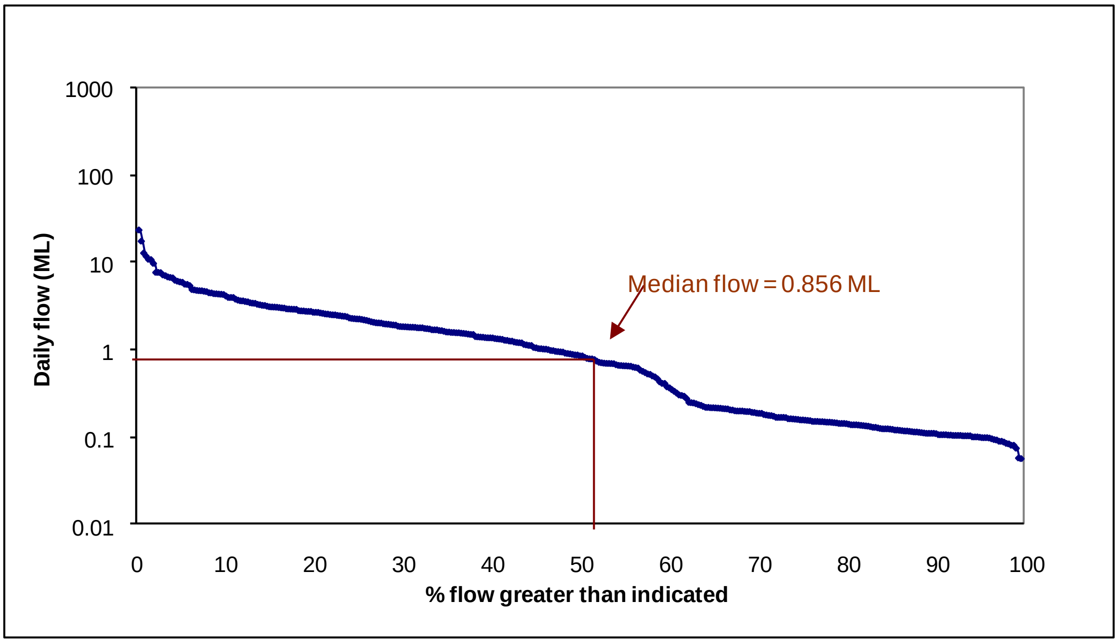


Figure 56. Flow frequency curve for station A5011007

Table 18. Characteristics of flow frequency

Flow Criteria	**% of Year**	**Days**
Cease to flow	–	–
Flow ≥ 1 ML/d	47	172
Flow ≥ 10 Ml/d	2	8
Flow ≥ 20 ML/d	1	4

F. REPORT ON FIELD SITE SOIL HYDRAULIC PROPERTIES

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Physical properties in selected soil pits in southern Fleurieu Peninsula

September 2009

Physical properties in selected soil pits in southern Fleurieu Peninsula, September 2009

Field

A number of sites were selected in the region with current land use, pine forest, native forest and pasture.

A pit was excavated at each site and the soil described (see associated report from PIRSA).

The infiltration rate of the surface soil was measured using three 50 cm diameter rings and the oblique ruler method for water depth measurement. For deeper soil layers, a flat surface was excavated on the edge of a trench above significant soil layers and an infiltration rate measured using two 30cm diameter rings. Several soils were too hard to insert rings so a small cavity was excavated, filled with water and the rate of infiltration measured as the water soaked in – using a vertical ruler.

At each significant layer, three undisturbed soil cores were taken. In some cases it was not possible to take the cores because the soil was too hard or brittle, in these cases soil aggregates were taken instead. The cores samples used thin walled techniques conforming to the Loveday criteria. The cores were 76mm diam x 50mm high.

Litter samples were taken in the pine forest and in the native forest and the water holding of that material measured.

A penetrometer with a 1cm diam tip, was used to measure the strength of the soil (or resistance to root penetration). In the surface layer (~0 - 40cm)

Laboratory

Undisturbed soil cores

The soil cores were weighed to establish initial (field sampled) water content and dry sand used to measure the volume of any cavities, so that the soil volume and hence density and porosity could be calculated. Some soils were not fully wet so they were placed on a tension plate and further water added to wet them up.

They were then placed on porous ceramic plates set to simulate a water table 5cm below the core base. They were then allowed to drain for12 hours and weighed again. This provides the water content at 0.5kPa tension.

The water table was then lowered to 100cm 'depth' - equivalent to a suction of 10kPa (take as field capacity) and the cores allowed 2 days to drain to reach equilibrium with the new water table before being weighed again. This process was repeated with water table at 600cm, (60kPa) allowing increasing time for equilibration (5 days for 600cm).

The cores were then oven dried to remove all water and weighed again. The volume, the dry weight of soil and the amount of water in the soil at each stage was then calculated along with such values as the density, total porosity, air filled porosity at field capacity (afp10).

Aggregates

Where the soil was too dry or brittle to take undisturbed core samples, a large aggregate was taken. This was used to measure soil density by the Saran method. Associated disturbed samples were used to measure the gravimetric water content at field capacity (10kPa) and wilting point (see method below). The gravimetric values were converted to volumetric (cc/cc) by multiplying by the density from the aggregates.

Wilting point
Small sub samples of disturbed soil were collected with the cores and used to measure water holding at higher suction levels. The loose soils were slurried onto a ceramic plate and subjected to air pressure of 1500kPa for 14 days, then removed and the water content measured.

These water content values measured as g/g were converted to a volume basis by using the density calculated from the core samples.

Infiltration rate in the field
Saturated infiltration rate was measured in the field using 3 x 50 cm rings on the surface soil and generally 2x 30cm rings on the control layer. Water flow was measured using a sloping ruler which gave a 10:1 magnification to water depth measurements. The control layer was selected as the layer most likely to be restricting from from the surface. To measure the control layer a pit (used for the soil description) was constructed with one sloping side (at about 30 degrees). A level surface was excavated in the slope for installation of the infiltrometer rings. Deeper layers were measured using a small excavation and a vertical ruler to measure water depth.

Spot measure of field water content

As part of the soil description, soil samples were collected at 20 cm intervals in each pit. The gravimetric water content of these was measured to provide a single point measure of the field water content at the time of sampling (10/8/2009 – 12/8/2009)

Water held in surface litter in native forest and pine plantation

The litter in the native forest was reasonably consistent in depth so an area adjacent to CH184 of 1960sq cm (50 cm diameter) was collected wet up and drained to 'field capacity' and its dry mass and water content measured.

The pine forest was extremely variable in litter cover – ranging from a depth of a few cm up to 40cm where there was tree thinnings to support it. No attempt was made to sample a specific area, but a sample was collected, wet up, and drained to measure its water holding capacity.

RESULTS

The full identification of the sites in given in table 1. Thereafter, the sites are referred to by the site number alone.

Table 1.   Site identification

Site number	Survey ID	Land use	Access/ ID	Comment
1	ch174	pasture	Wither Swamp	hilltop
2	ch175	pasture	Wither Swamp	Mid slope
3	ch176	pasture	Wither Swamp	Lower slope
4	ch177	swamp	Wither Swamp	swamp
5	ch178	Pine forest	Foggy farm	Upper slope
6	ch179	Pine forest	Foggy farm	Mid slope
7	ch181	Pine forest	Foggy farm	swamp
8	ch182	Pine forest	Foggy farm	swamp
9	ch180	Pine forest	Foggy farm	Lower slope
10	ch183	Native forest	Springs Rd	hilltop
11	ch184	Native forest	Springs Rd	Mid slope
12	ch185	Native forest	Springs Rd	Lower slope
13	ch186	Native forest	Springs Rd	swamp

Table 2. Results of infiltration tests

site	survey site ID	surface infiltration mm/min	control layer		C horizon infiltration mm/min
			depth cm	infiltration mm/min	
1	ch174	1.75	30	7.04	0.040
2	ch175	2.95	30	5.14	1.760
3	ch176	1.47			underwater
4	ch177	0	40	0.03	n/a
5	ch178	1.22	40	0.4	n/a
6	ch179	2.51	50	0.89	0.030
9	ch180	8.64	40	0.78	0.002
7	ch181	0.1	30	0.04	0.001
8	ch182	14.2	40	1.27	0.080
10	ch183	40	40	0.75	0.010
11	ch184	9.8	40	6.5	0.070
12	ch185	3.94	50	1.33	0.010
13	ch186	0	n/a	saturated	underwater

Table 4. Penetrometer strength in field. Note that values over 3MPa would substantially reduce root growth, values over 5 MPa would prevent root growth except in cracks.

Site	Survey site ID	Penetrometer Strength Mpa		
		10cm	20cm	30cm
1	ch174	n/a		
2	ch175	***3.9***	***4.1***	***6.1***
3	ch176	***1.5***	***3.0***	***6.1***
4	ch177	***3.7***	***6.4***	***7.1***
5	ch178	***2.6***	***4.6***	***4.6***
6	ch179	***3.9***	***6.8***	
9	ch180	n/a		
7	ch181	***2.3***	***3.4***	***3.4***
8	ch182	***2.7***	***5.5***	***7.1***
10	ch183	***1.7***	***2.1***	***4.0***
11	ch184	***1.7***	***4.6***	***4.5***
12	ch185	***4.9***	***2.7***	***7.1***
13	ch186	n/a		

Table 3. Water Release from aggregates and undisturbed core samples

Site	Survey ID	Depth cm	Field water	Volumetric water content(cc/cc) 0.05kPa	10kPa	60kPa	1500kPa	Density g/cc	Total Porosity
1	ch174	0	0.46	0.49	0.42	0.35	0.08	1.00	0.62
		30	0.30	0.37	0.33	0.29	0.22	1.38	0.48
		80			0.40		0.22	1.58	0.40
2	ch175	0	0.49	0.50	0.45	0.39	0.11	1.01	0.62
		30	0.29	0.34	0.31	0.28	0.19	1.33	0.50
		80	0.25	0.31	0.29	0.27	0.21	1.02	0.62
3	ch176	0	0.53	0.54	0.46	0.38	0.10	1.03	0.61
		40	0.29	0.32	0.28	0.25	0.19	1.05	0.60
		80			0.39		0.29	1.65	0.38
4	ch177	0			0.45		0.09	1.46	0.45
		20			0.40		0.34	1.60	
		40			0.20		0.06	1.86	0.30
		80			0.31		0.34	1.81	0.32
5	ch178	0	0.33	0.42	0.30	0.26	0.07	0.88	0.67
pines		40	0.23	0.30	0.24	0.21	0.14	0.98	0.63
		80			0.31		0.36	2.26	0.15
6	ch179	0	0.49	0.53	0.36	0.31	0.05	0.56	0.79
		50	0.23	0.29	0.25	0.22	0.15	1.06	0.60
		80			0.25		0.14	1.59	0.40
9	ch180	0	0.32	0.44	0.31	0.26	0.13	0.55	0.79
		40	0.36	0.44	0.39	0.35	0.26	1.27	0.52
		130			0.32		0.15	1.51	0.43
7	ch181	0	0.64	0.64	0.54	0.43	0.11	0.74	0.72
swamp		20	0.32	0.36	0.31	0.25	0.08	1.67	0.37
		60			0.33		0.19	1.67	
		80			0.35		0.27	1.67	
		100			0.35		0.27	1.67	0.37
8	ch182	0	0.53	0.61	0.48	0.44	0.16	0.46	0.83
Swamp		40			0.61		0.35	1.27	0.52
		80			0.62		0.30	1.26	0.53
10	ch183	0	0.41	0.42	0.33	0.29	0.16	0.80	0.70
scrub		40	0.23	0.28	0.23	0.19	0.12	1.21	0.55
		80			0.29		0.25	2.00	0.25
11	ch184	0	0.31	0.44	0.31	0.28	0.08	0.67	0.75
scrub		40	0.37	0.44	0.41	0.38	0.30	1.29	0.51
		120			0.30		0.19	1.59	0.40
12	ch185	0	0.28	0.41	0.29	0.27	0.09	0.71	0.73
scrub		50	0.34	0.40	0.37	0.35	0.29	1.36	0.48
		120			0.38		0.29	1.55	0.41
13	ch186	0	0.69	0.68	0.87	0.48	0.17	0.63	0.76
swamp		40	0.37	0.40	0.38	0.35	0.29	0.90	0.66
		120			0.39		0.35	1.63	0.39

Table 4: Spot water content of field (note these values are in g/g and will need to be multiplied by the soil density at each depth and the depth over which they apply to get the water held in the soil at the time of sampling (12/10/2009)

Site	depth	water g/g		Site	depth	water g/g
ch174	1	0.315		ch181	1	0.430
	2	0.146			2	0.210
	3	0.239			3	0.182
	4	0.167			4	0.200
	5	0.160			5	0.236
ch175	1	0.169		ch182	1	0.987
	2	0.230			2	0.638
	3	0.286			3	0.511
	4	0.325			4	0.268
	5	0.267			5	0.372
	6	0.240			6	0.190
	7	0.214				
ch176	1	0.491		ch183	1	0.437
	2	0.211			2	0.142
	3	0.356			3	0.167
	4	0.217			4	0.218
	5	0.310				
ch177	1	0.624		ch184	1	0.235
	2	0.193			2	0.214
	3	0.292			3	0.271
	4	0.220			4	0.215
					5	0.253
ch178	1	0.184			6	0.117
	2	0.124				
	3	0.171				
	4	0.201				
	5	0.273				
ch179	1	0.506		ch185	1	0.412
	2	0.212			2	0.304
	3	0.211			3	0.186
	4	0.202			4	0.199
	5	0.154			5	0.239
					6	0.235
ch180	1	0.873		ch186	1	1.192
	2	0.277			2	0.782
	3	0.219			3	0.562
	4	0.232			4	0.398
	5	0.124			5	0.340
					6	0.273

Water holding of Surface litter in native forest and pine plantation

The sampled site in the native forest had 952 g/sq m of dry matter which held 0.6cc water per gm of dry matter at field capacity. Pine litter held 2.7cc water per gm of dry litter at field capacity.

G. FIELD SITE INSTRUMENTATION

Figure 57. Springs Road Native Forest Reserve Field sites

Springs Road was instrumented as shown in Figure 57 with streamflow out (not shown, but adjacent to the 'rain gauge' location); rainfall ('pluviometer;); soil moisture monitoring ('enviroscan'; 'access tube') and upstream and downstream continuous groundwater observation wells ('piezometer'). Flow is from right to left.

Figure 58. Foggy Farm field site showing instrumentation locations

Foggy Farm was instrumented as shown in Figure 58 with streamflow in and out of the wetland (Qin ; Qout); continuous groundwater monitoring in the wetland (FF01:FF04); soil moisture (PR01:PR07, ESFF); rainfall, dam level and pumped offtake (RF).

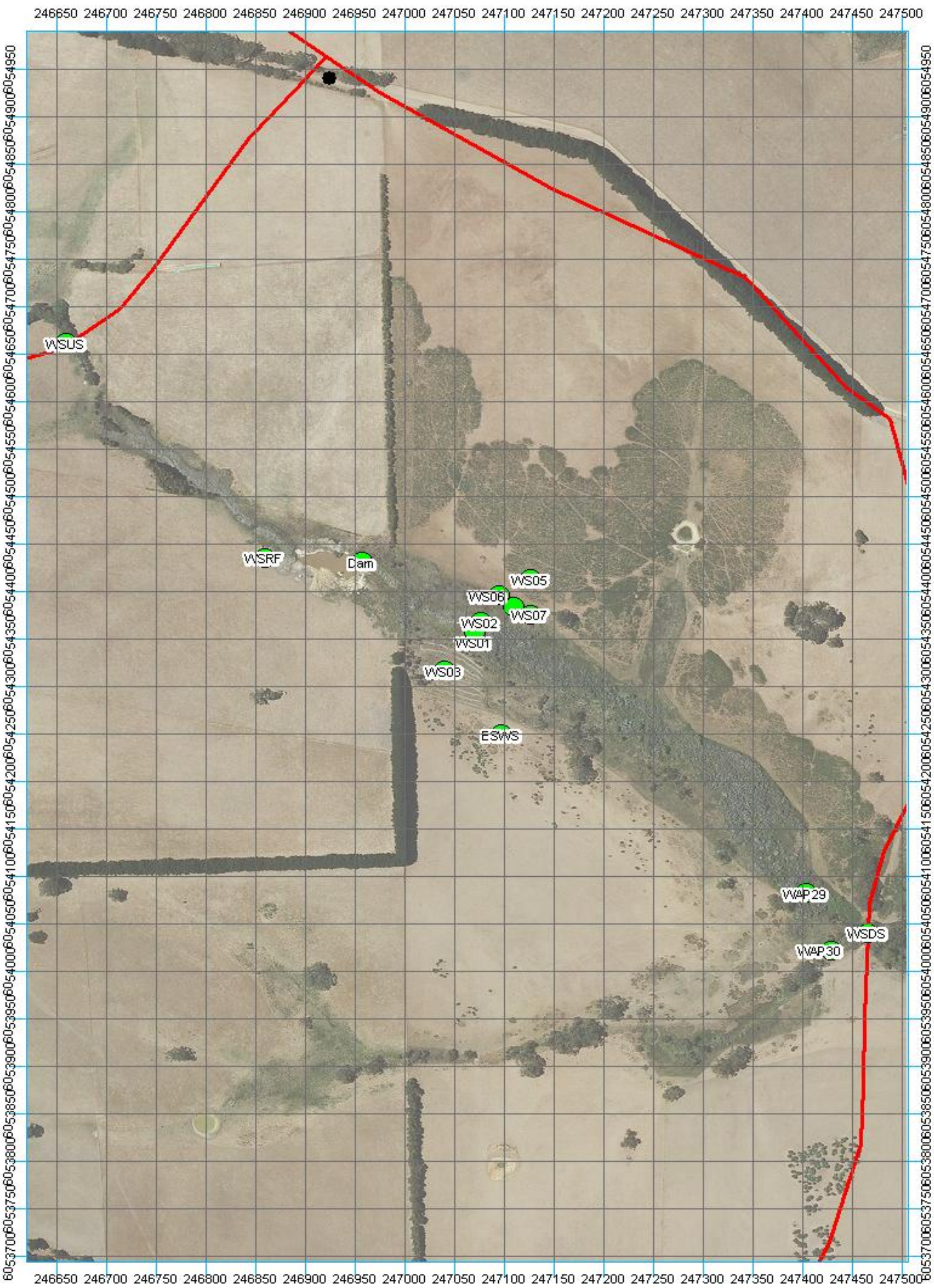


Figure 59. **Wither Swamp field site instrumentation**

Wither Swamp was fitted as shown in Figure 59 with streamflow in and out (WSUS; WSDS), groundwater wells WAP29 and WAP 30 at the downstream end and a transect WS1-7 recording levels within the local flow system and adjacent wetland; a pluviometer (WSRF) and dam level (Dam); continuous soil moisture (ESWS).

H. VEGETATION COMMUNITY AT THE THREE FIELD SITES

Baseline surveys consisting of multiple transects were undertaken at each site to characterise vegetation cover and abundance at each site. Where possible, sites were co-located with groundwater observation wells to provide an indication of the water regime at each point in the wetland for analysis, other sites were located to cover the wetland extent with at least one site at the lower, middle and upper reaches of each site. Data were collected as cover classes in a modified Braun-Blanquet scoring system with life stage data (vegetative, flowering, fruiting, seed, senescence) also recorded.

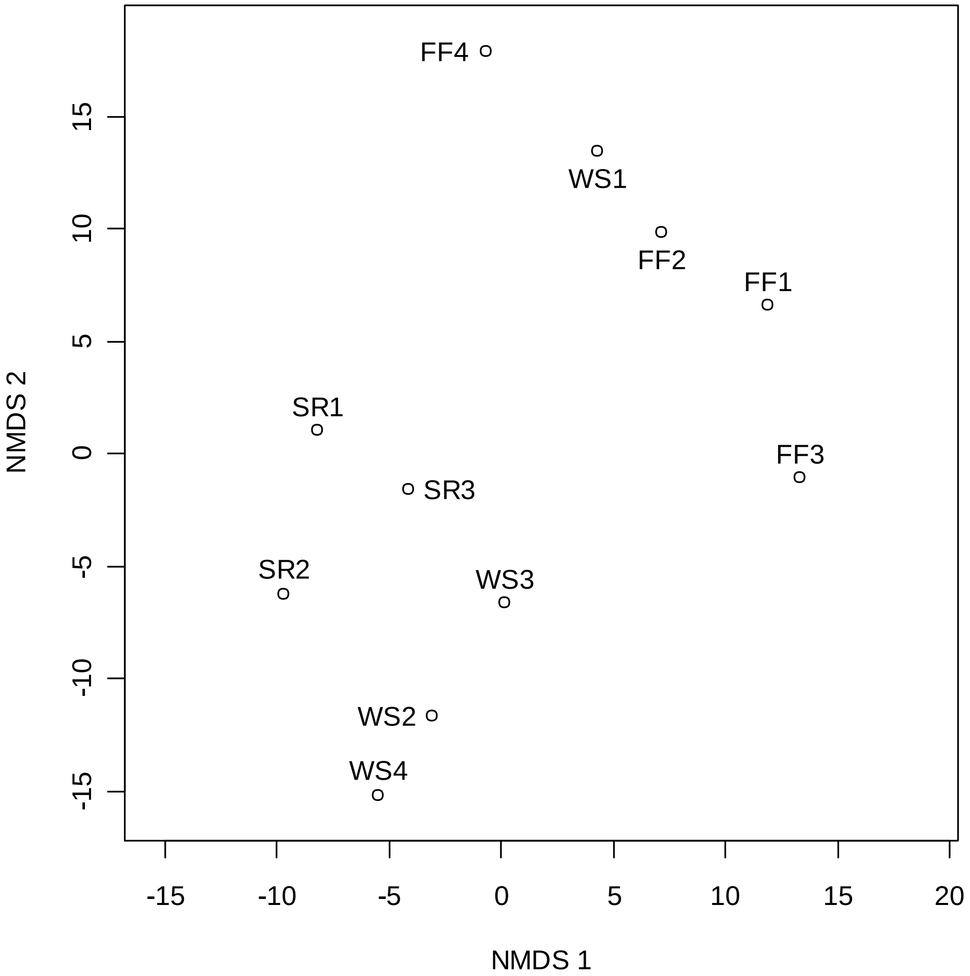
A total of 74 species were recorded across all sites. Total species richness for individual wetlands was Springs Road 41; Foggy Farm 48; Wither Swamp 61. Table 19 presents summary data for each different transect at each site. The overall difference in this community state variable is in part attributable to differences in wetland area, but hydrology appears to perform a key structuring role (see multivariate analyses below).

Table 19. Relative location and species richness data – all sites

Site code	Location	Chainage	Length	Species richness
FF1	upstream end of wetland	630	29	19
FF2	lower middle reach	320	94	16
FF3	upper middle reach	500	60	35
FF4	downstream end of wetland	240	81	19
SR1	downstream end of wetland	160	22	25
SR2	middle reach	560	40	14
SR3	upstream end of wetland	890	28	21
WS1	downstream end of wetland	70	42	22
WS2	lower middle reach	300	90	37
WS3	upstream end of wetland	550	77	32
WS4	upper middle reach	430	70	37

FF = Foggy Farm; SR = Springs Road; WS = Wither Swamp

In order to compare vegetation associations across the three sites, species were converted to their respective vegetation functional groupings (Casanova *et al.* unpub.). Data were assigned a percentage value corresponding to the mid-point of the cover class and arcsine square root transformed to provide an improved distribution of values. A distance matrix was then calculated using the Bray-Curtis distance measure (Bray and Curtis 1957). The distance matrix was used as the basis for ordination and classification. Multivariate analyses were undertaken using the programming language R (R Development Core Team). Ordination was non-metric multidimensional scaling (NMDS) using the 'labdsv' package (Roberts 2007), while the classification was undertaken using Unweighted Pair-Group Method with Arithmetric mean (UPGMA) in package 'stats' (R Development Core Team).



Site codes: WS = Wither Swamp; SR = Springs Road; FF = Foggy Farm. Numeric codes indicate location according to Table 19

Figure 60. NMDS ordination (stress = 0.06) of vegetation transects from the three study sites with species converted to plant functional group and pooled for each transect

Transects were generally closest in functional group proportions to others within the same wetland. The exception to this was transect WS04 from the Wither Swamp, which grouped with two of the Foggy Farm transects. This transect was not actually located within the wetland, but traversed the watercourse downstream where bank elevation was up to three metres above the watercourse and riparian vegetation was dominated by bracken.

Grouping patterns were consistent between the NMDS and UPGMA clustering, with NMDS axis 2 more effective at separating the different transects and appears to be highly correlated with water availability based on field observations on relative water availability at each site (that is, the driest sites are located at the top of the figure and the permanently wet sites are associated with low scores on Axis 2).

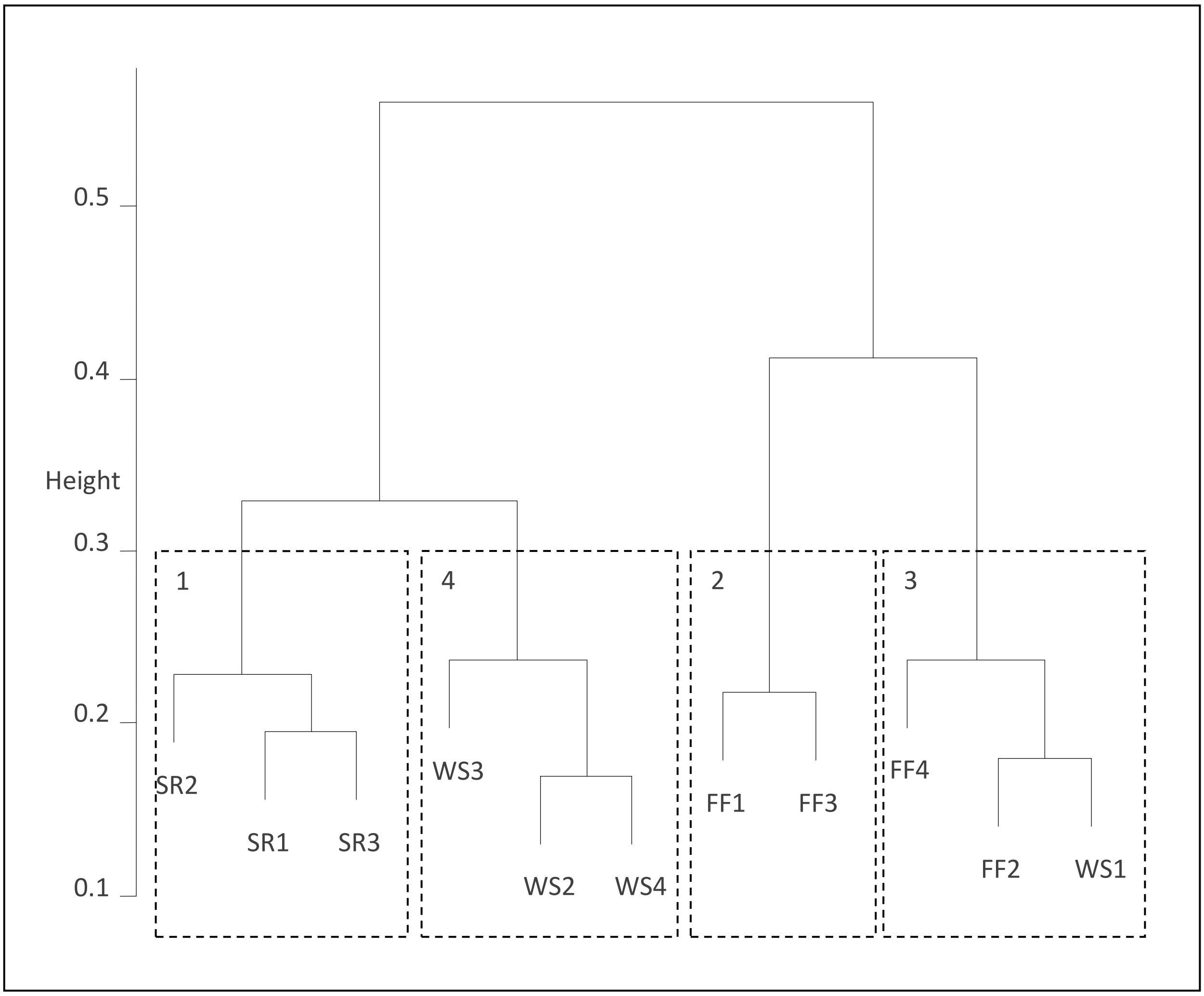


Figure 61. UPGMA clustering of site transects, grouping as shown in boxes. Data as for Figure 60.

The cluster produced four clear groupings, at height 0.3, which were subjected to Dufrene-Legendre Indicator Species analysis (Dufrene and Legendre 1997), as implemented in the labdsv package (Roberts 2007). This analysis more clearly indicates the plant functional groups that contribute most to the differences between the associations represented in the clustering.

Results of the Indicator species analysis are summarised in Table 20, which presents the relative abundance of the different vegetation functional groups in each cluster grouping. The relative water requirement for the different functional groups decreases from top to bottom of the table. With reference to Table 20 and aquatic and semi-aquatic plant functional group (PFG) classification as presented in Vanlaarhoven and van der Wielen (2009), the following is noted:

Group 1 does not have any statistically significant proportional contribution from any PFG. High occurrence of the semi-aquatic groups Amphibious fluctuation tolerant low-growing (Aftl), Amphibious fluctuation tolerant emergent (Afte) and Terrestrial damp (Tdamp) are observed. Given the high proportion of Aftl present in this group, only the low overall prevalence of the group would have prevented this being statistically significant.

Group 2 is distinguished by a statistically significant occurrence of the semi-aquatic functional group Aftw (fluctuation tolerant woody growth habit). Proportions of Aftw are relatively consistent across the other Groups. Submerged functional types were totally absent from this group.

Group 3 is characterised by a significant cover (51%; $p < 0.01$) of Tdry (terrestrial species) and low proportions of water dependent functional groups.

Group 4 represents the most unique habitat based on the number of indicator species. The grouping, comprising three transects from the Wither Swamp is characterised by significantly high ($p < 0.01$) proportions of the true aquatic groups submerged emergents (SE – 93% of total occurrence); and amphibious fluctuation responders with plastic growth habit (Afrp, 70% of total occurrence) are significantly associated with this group at the 5% level. Semi-aquatic Afte (typically sedges; 47% of total occurrence) and Tdamp functional group (61% of total occurrence are also significant indicators of the group ($p < 0.01$).

Table 20. Relative abundance of different vegetation functional groups within each clustering group across all transects. True aquatics are shown above the dotted line

PFG[1]	Group number from UPGMA clusters[2]			
	1	2	3	4
Submerged emergent	3%	0%	4%	93%**
Amphibious fluctuation responder plastic	6%	11%	13%	70%*
Amphibious fluctuation tolerant emergent	35%	13%	6%	47%**
Amphibious fluctuation tolerant low-growth	65%	12%	4%	20%
Amphibious fluctuation tolerant woody	13%	49%*	20%	18%
Terrestrial damp	21%	6%	11%	61%**
Terrestrial dry	16%	23%	51%**	10%

1. Plant functional group, an indication of water dependence. Rows indicate roughly increasing water requirements from top to bottom. Detail on each PFG can be found in Vanlaarhoven and van der Wielen (2009).
2. Asterisks indicate statistically significant results (* $p < 0.05$; ** $p < 0.01$).

With reference to the dendrogram in Figure 61, the highest split (height ~ 0.55) is between habitat favouring terrestrial or woody semi-aquatics and semi-aquatic or aquatic favoured habitat. There is a much greater separation in these plant communities at functional group level, than between the semi-aquatic (Group 1 - Springs Road) and aquatic (Group 4 - Wither Swamp) communities.

While the Wither Swamp transects remained permanently saturated at the surface, it is of interest that terrestrial damp species are so dominant within these transects. This may reflect the general range of suitability for wetland fringe areas owing to the permanent saturation. Alternatively this may reflect the interaction of land use with the dynamics of wetland ecotones where Tdamp would be expected to be common. The Wither Swamp comprises a fenced buffer area formerly under pasture. In the absence of the original woody terrestrial vegetation Tdamp species may currently dominate the ecotone. Although Springs Road is arguably more suitable habitat for Tdamp PFG species, the site is remnant vegetation and the more even dominance of Tdry and Tdamp may reflect the composition of wetland ecotones under pre-clearance conditions. Foggy Farm, being a commercial plantation likely planted areas that may originally have provided optimal Tdamp habitat.

The groupings and conclusions discussed above are consistent with watertable dynamics for 2009 (Fig. 22), where Foggy Farm dried completely during autumn in 2008 and 2009, while Springs Road watertables fell to greater than one metre below ground level. Wither Swamp remains permanently saturated with watertables within a few centimetres of the surface. Findings are also consistent with the general conclusions about the range of functional groups suited to perched wetland habitat in

Section 3.2. Wetlands supporting high proportions of true aquatic species are relatively rare in the landscape and therefore of high conservation value.

I. VOLUMETRIC SOIL MOISTURE DATA AND INTERPRETATION

SOIL MOISTURE DATA

Volumetric soil moisture (VSM) monitoring was installed at all three sites to observe soil profile wetting and drying processes within both wetlands and catchments. The method to measure soil moisture was frequency domain reflectometry (FDR) and the instrument used was the *Diviner 2000*, manufactured by Sentek Instruments (Stepney, South Australia). Instruments require 50 mm PVC access tubes that are installed via a purpose built tripod auger system that ensures vertical orientation and also minimises the possibility of air gaps between the tube and the soil. The location of access tubes where measurements were taken is shown in Figures 57-59.

The *Diviner 2000* is manually read by inserting and extracting the transducer along the access tube, providing a single point in time soil moisture reading of the whole soil profile. VSM readings are taken at 10 cm intervals providing depth based and integrated measures of VSM in the vicinity of the access tube from 5 cm to 165 cm depth. The observed readings at each depth represent mm of water per 10 cm of soil (Sentek 2007). For the purposes of this study data was collected from soil moisture access tubes on an approximately monthly basis over 2009.

Monitoring effort across the three sites was not consistent owing to the logistics of installing the tubes. At Foggy Farm, owing to the dry conditions, four access tubes were able to be installed as a longitudinal transect within the wetland itself prior to February 2009. Soil tubes were located adjacent to the four piezometers reading saturated conditions at the site (Fig. 58). Three additional tubes were placed around the catchment outside the wetland across low and middle slope locations of interest. Full depth installation at some sites was not possible owing to shallow occurrence of bedrock. At Wither Swamp and Springs Road wetlands, wetter conditions prevented the level of instrumentation possible at Foggy Farm, though single access tubes were placed adjacent to wetland sediments at both sites, but these were not installed until spring 2009.

Volumetric soil moisture content measured via FDR is greatly influenced by soil texture and can only provide an absolute measure if complex calibration procedures are undertaken at each site. This was beyond available resources. Changes between two readings at the same site and depth present a reliable indication of the relative change in the volume of water stored in the profile. Changes in soil moisture between monthly readings have provided the most useful information in interpreting wetland soil moisture dynamics.

Soil water content dynamics – Foggy Farm wetland

The methods used below to display the data herein are either soil moisture, or change in soil moisture profiles. Both of these charts represent the soil profile from 0 to 1.6 metres on the vertical axis. The horizontal axis shows either volumetric soil moisture (Fig. 62), or the change in VSM over the month since the prior measurement (Fig. 63). For a VSM change profile, a vertical line passing through zero at all depths would indicate no change in soil moisture between monthly readings. Increases in soil moisture will appear on the right of a vertical line passing through zero (positive values) and decreases in soil moisture plot to the left. Best understanding is generated with reference to both Figures.

Foggy Farm access tube PR03 over 2009 was located in the central wetland area, within a stand of tea-tree (*Leptospermum* spp.) as shown in Figure 58. The soil profile at this location (Foggy Farm soil description site 87) has a 25 cm thickness topsoil comprising 10 cm loamy peat and 15 cm dark coloured loam; this is underlain by a subsoil of dark grey, sodic (dispersive) light clay (upper subsoil to 45 cm) and silty light clay to a depth of 115 cm; this in turn is underlain by grey sodic clay layers (to 160 cm); highly weathered basement rock becomes clearly evident at a depth of 300 cm. Most roots occur in the upper 45 cm, while significant quantities extend to 175 cm and minor quantities were observed to 300 cm;

none were observed in the layer below this. (Below 280 cm are two distinct soil substrate layers showing evidence of preferential throughflow around soil structural aggregates.) Between 115 and 140 cm is a layer of clay with heavier texture than layers above or below, as well as a considerably lower pH. As indicated in Figure 62 and 63 a change in soil moisture dynamics is evident between 110 and 120 cm. This supports the field observation of different hydrological characteristics to adjacent layers and the layer may act like an aquitard constraining the vertical movement of water.

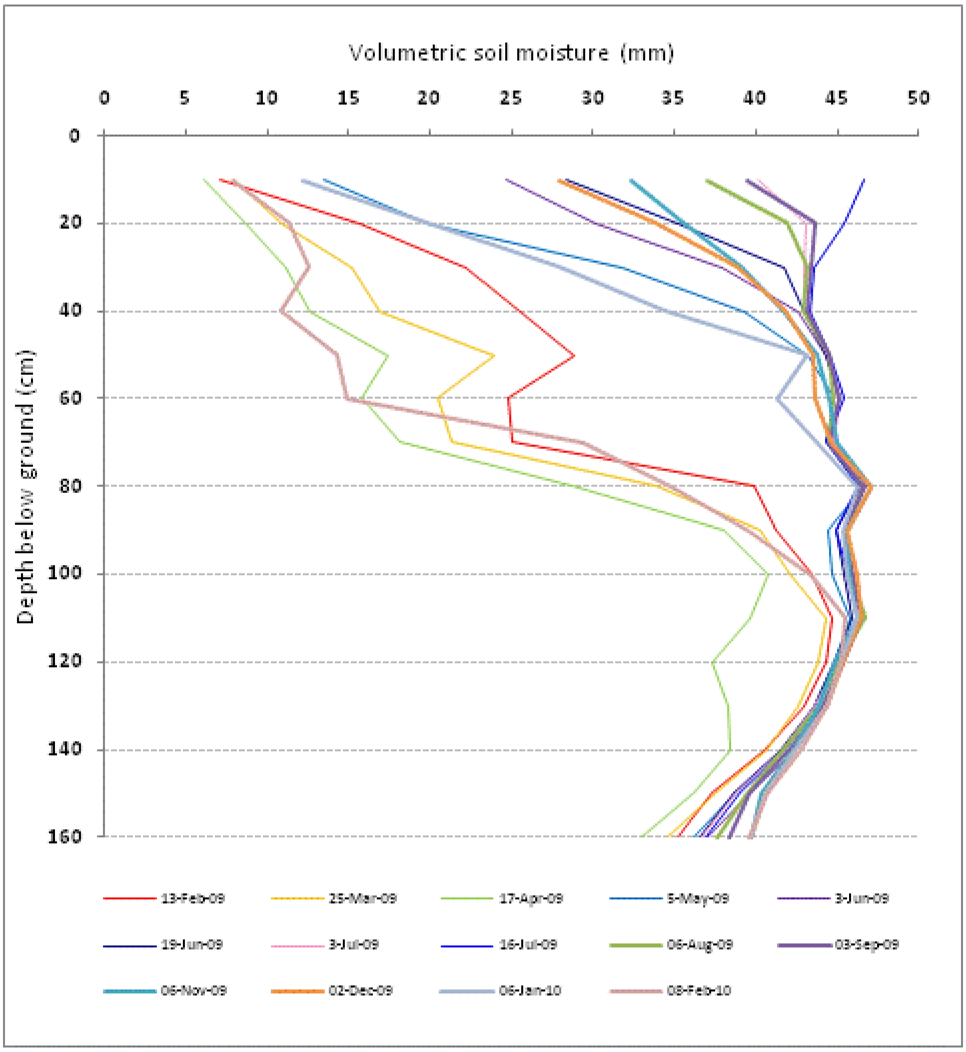


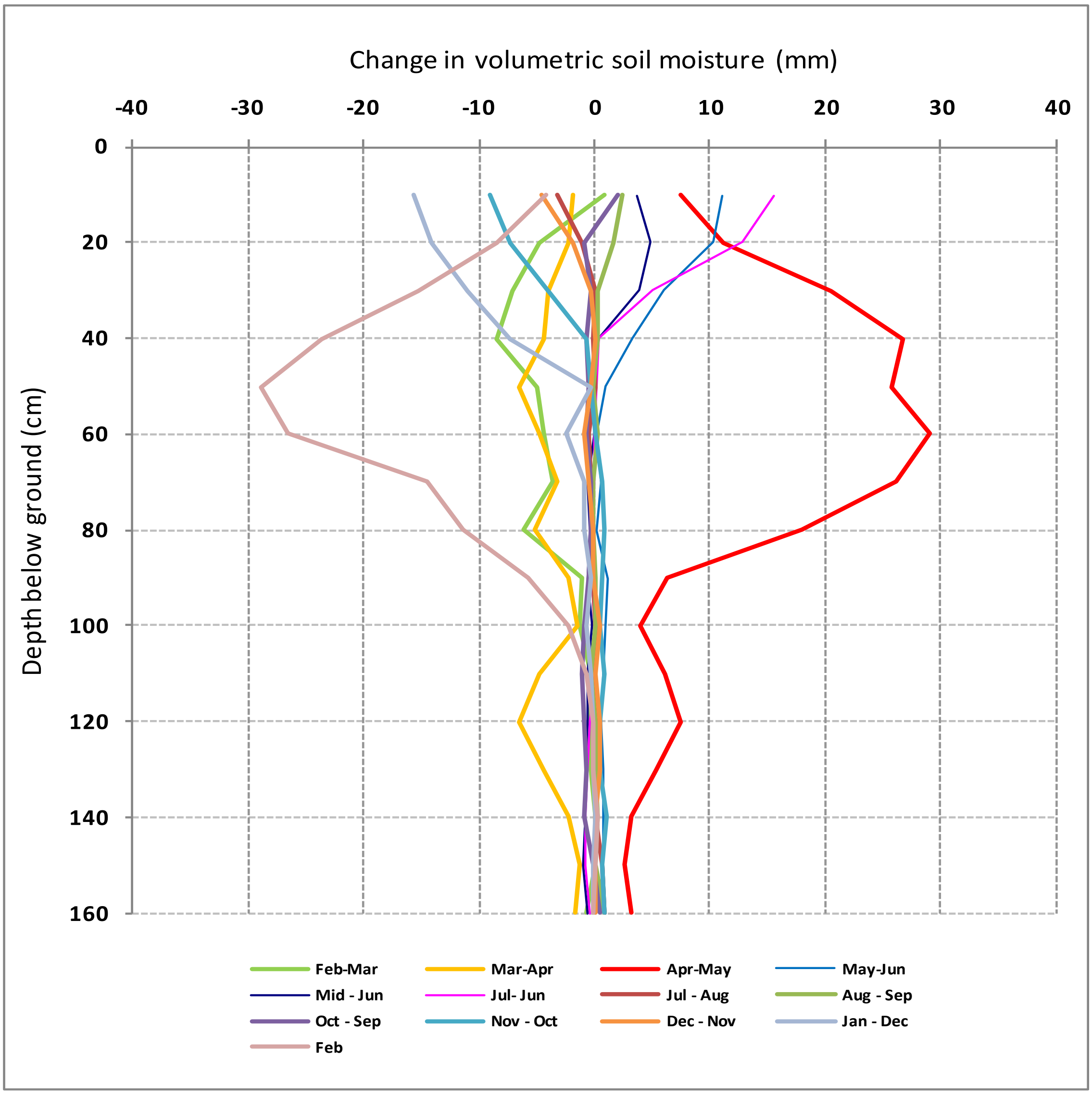
Figure 62. Monthly time series of volumetric soil moisture – Foggy Farm wetland

The soil moisture profile (Fig. 62) shows the monthly measured water content, with each line representing a monthly reading. The value 0 on the vertical axis represents ground level and other values represent the depth of soil which the reading corresponds to. The soil moisture value is read from the horizontal axis. A sequential drying of the wetland soil profile is indicated by the movement to the left of the figure from 13 Feb (red line), through 25 Mar (yellow line) and 17 Apr (green). Between the 17 April reading and the next monthly reading 5 May (blue line) significant break of season rains fell. A total of 133 mm fell between 17 April and 5 May and the difference between the two lines indicates the increase in soil water storage as a result. By summing the difference across all depths it is indicated that soil moisture stores increased by 124 mm. This suggests that the change in moisture is attributable almost entirely to the vertical infiltration of rainfall and little, if any, concentration of water via surface, or sub-surface movement is indicated.

The changes described above are also evident in the change profile for the same data (Fig. 63). In this profile, the lines represent the monthly readings in Figure 62, which are subtracted in time order, such that the March reading is subtracted from the February reading and so on. By this means the change in soil moisture storage held (mm) can be compared at each site in response to the depth of rainfall received (mm).

As with Figure 62, sequential drying of the profile can be seen in the difference between Feb-Mar trace (green) and the Mar-Apr trace (yellow). In the earlier trace water is removed from the top part of the profile from evapotranspiration, but water content at depth is maintained by the continued flow of moisture through cracks within the subsoil clay (pathway 2 Fig. 23). By April however the water being removed is no longer able to be replenished and the remaining moisture within the preferential pathways has been removed.

The red trace (Apr–May) indicates the changes across the profile to the significant depth of rainfall received in the period. Effectively all soil stores are replenished over this single month and additional inflows over the winter-spring serve only to saturate surface soils, the most rapidly draining. The restoration of saturation within subsoils can be seen as the mirror image of the yellow trace at around 1.2 metres. For the remainder of the winter-spring period, moisture changes are only observed in the top 40 cm. These readings indicate the relative saturation of the peat-loam layer at the surface on the date of measurement.



Readings in this profile represent the arithmetic difference between consecutive readings in Figure 62. That is, the reading at each depth for the later month is subtracted from the reading at the same depth the month prior, with this value displayed. Hence a zero change in soil moisture over the month would show as a vertical line through 0 at all depths. Positive values on the horizontal axis represent an increase in water held in the profile and negative values indicate a loss of water held.

Figure 63. Monthly changes in volumetric soil moisture – Foggy Farm access tube PR03

In addition to soil water content responses to specific rainfall periods, considerable direct and derived understanding of general behaviour was possible through inspection of soil water content profiles such as these. It is particularly instructive to reference changes in soil water content to knowledge of the soil structure and saturated water movement pathways described in Section 5.2. Notable observations include an estimate of extractable water can be obtained through a sum of the difference in maximum and minimum water content readings across the entire time series at each depth. This simple analysis indicates 75% of the extractable soil water was in the top 70 cm of the profile. This represents water content variation in the organic layers overlying the weathered basement. The large variations within this layer are due both to the high porosity soil texture and plant water dependence of the surface soils. At this profile the maximum soil moisture variation occurs in the top 80 cm deep, which represents an indicative maximum depth observed at the site for wetland organic surface layers.

Soil profiles in these organic layers tend to exhibit 'wet' and 'dry domains' and rapidly transition between the two states. Most of the wet season moisture profiles are very close in value and limited change occurs. This is the period where soil moisture content is maintained between field capacity and saturation. Once the supply of water from pathways 1 and 2 ceases, evapotranspiration demand rapidly depletes soil water – this is evident in the difference in profiles between 2 Jan 2010 and 8 February 2010, where soils move to the 'dry domain'.

The difference between the 2 Jan and 8 Feb soil water content readings (-141 mm) with added rainfall (29 mm) represents an estimate of soil evaporation and transpiration demand for tea-tree vegetation at the site, yielding a value of 170 mm, or 5.2 mm/d. As profiles are clearly not water limited, this is a reasonable estimate of optimal demand. During the period 13 Feb–17 April, the two equivalent estimates for site PR03 were 2.3 and 2.6 mm/d indicating a possible soil moisture constraint on evapotranspiration, although other climatic factors such as solar angle (a determinant of incident radiation) would also contribute to reduced demand by April. With existing information it is not possible to determine the relative contribution by evaporation from the soil surface and whether tea-tree access to transpirable water was limited or not. This was a major intention behind the installation of the Class A pan evaporimeter at Parawa, however instrument failure meant this was not possible.

At depths below 110 cm there is little change in moisture levels from the wettest to the driest reading. This part of the profile represents the saturation level in the cracks and macropores associated with throughflow in the wetland B horizon (pathway 2, Sect. 5.2). As can be seen from around 90 cm, the difference between wettest and driest parts of the profile is around 10 mm, consistent with the piezometer data derived estimates of secondary porosity of 0.1. This is the residual store used in modelling the catchments, which does not drain completely and must fill prior to runoff commencing from the wetland.

Soils fill consistently across the profile once rainfall commences, but dry more gradually at depth. The deepest soil (associated with pathway 2) is the last to dry, but wets up immediately following a reasonable rainfall excess. This indicates the rapid vertical infiltration of water within wetlands through the surface layers, consistent with the groundwater hydrographs at the seasonal sites. As soil storage in the lower profile is small and effectively limited to the secondary porosity, this saturates as soon as reasonable breaking rains commence. This is clearly evident in the change profile (Fig. 63), as the red profile which shows the responses across the profile to the first breaking rain. Subsequent rainfall does not increase water content in this layer, suggesting saturation occurred early in the monitoring season.

In contrast surface layers continue to increase in water content as saturated flow starts to occur from the surrounding catchment, accumulating in the wetland. By July (blue trace in Fig. 62 – the only soil reading taken on a day where surface saturation was widespread in the wetland) surface saturation is evident and streamflow had commenced.

During winter spring the only observed changes in moisture levels represent surface loams filling and draining in response to rainfall. This rapid filling and draining in response to winter-spring rainfall events can be seen in the top 20 cm of the change profile (which approximately corresponds to topsoil thickness); for example the difference between the two June readings leading to the blue mid-June profile was saturation of the surface loams. Other than the 20–40 cm layer, winter-spring profiles remained at similar water contents. This is also evident in the piezometer data, where surface soil saturation was not persistent at the site (Fig. 22, Sect. 5).

The patterns observed at PR03 in the Foggy Farm wetland presented above were consistent across all Foggy Farm wetland soil moisture sites. At the other two field sites, variations in soil water content within wetland sediments was not observed to the same degree as Foggy Farm. Soil access tubes in wetlands at the other two sites were not deployed until spring 2009, but change profiles for Springs Road and Wither Swamp wetlands are also presented for the period September 2009–March 2010.

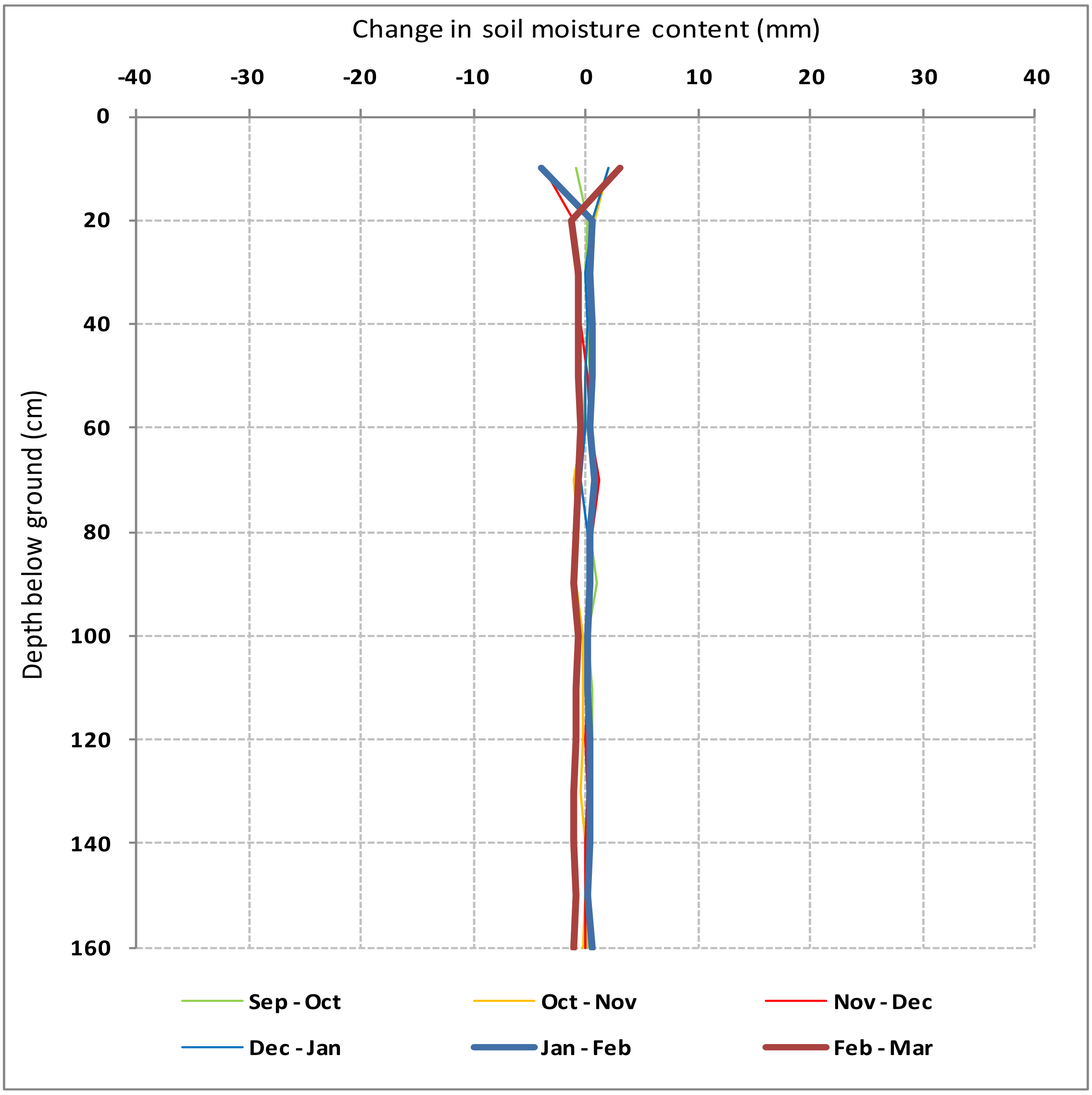


Figure 64. Monthly change profile for soil moisture content – Wither Swamp (PR23 adjacent to WS07 in Fig. 59)

Change profiles for Wither Swamp (Fig. 64) and Springs Road (Fig. 65) indicate almost no variation in moisture content across the wetland soil profile. The Wither Swamp site was adjacent to the wetland within pasture grasses on the wetland fringe and some drying of surface layers associated with pasture root depths can be seen. Springs Road was adjacent to the wetland in a stand of native deep rooted perennial shrubs and shows a greater soil water variation likely to be due to plant transpiration. Neither figure suggests changes to the state of soil water content at depth.

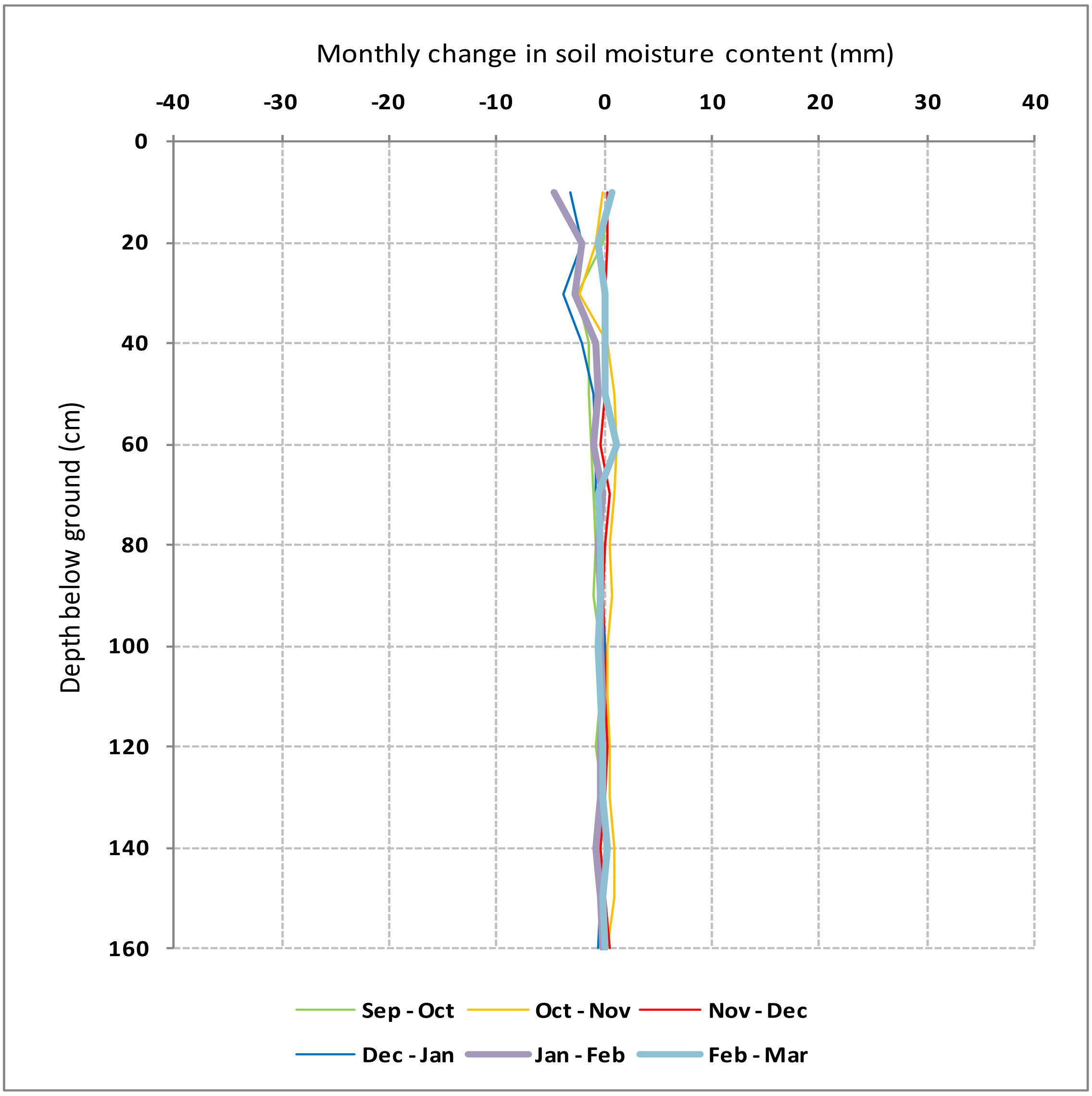


Figure 65. Monthly change profile for soil moisture content, Springs Road (PR31 – denoted access tube in Fig. 57)

Continuous Soil moisture monitoring data

In contrast with the Diviner 2000, the Enviroscan logged hourly variation in soil moisture content at four depths. In all field sites depths were 10, 30, 90 and 150 cm. Instruments at the plantation and forest study catchments were located at comparable positions on lower slopes, however the Wither Swamp instrument was in a lower position on a concave section of the foot slopes that became seasonally waterlogged.

Differential texture responses of the FDR soil moisture measurement method (see above) mean that direct comparisons of absolute moisture content (the vertical axis) across instruments even within the same depth are not valid. As with the Diviner 2000 data, comparison of changes in moisture content for a single depth can be read as mm/10 cm (that is %) change in soil moisture at the depth indicated ±5 cm.

The access tubes for all soil water monitoring may have been subject to water flowing via the outside of the tube from higher saturated points in the profile giving elevated readings at lower soil depths. However the data present some insights into broad soil profile moisture dynamics at the different sites.

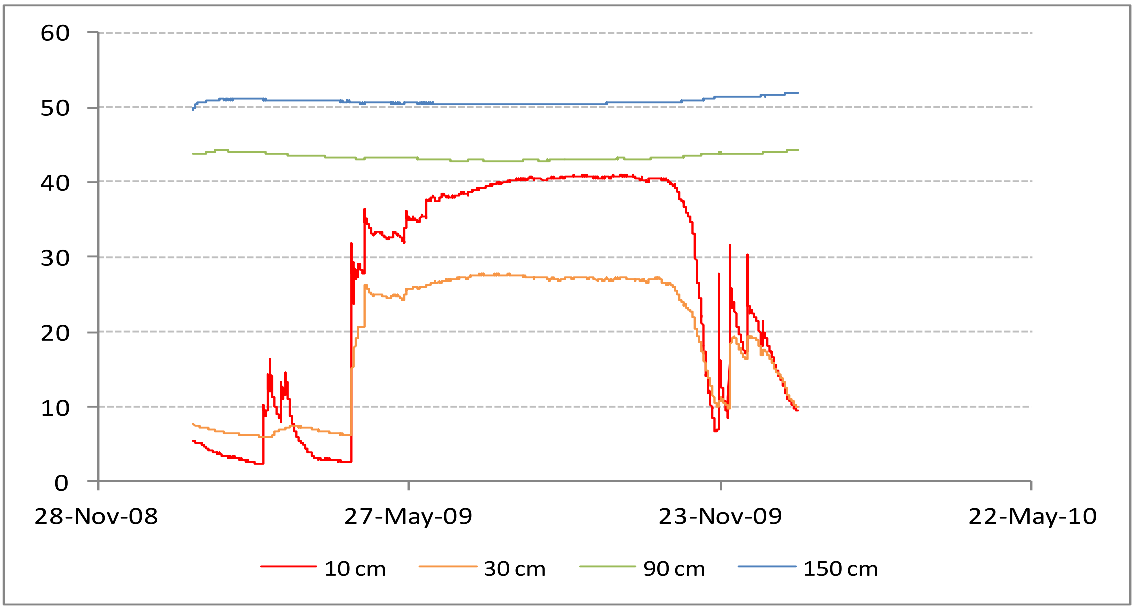


Figure 66. **Continuous volumetric soil moisture at four depths – Wither Swamp. Vertical axis indicates volumetric water content (mm/10 cm) held in the soil profile ±50 mm of sensor depth.**

The Wither Swamp continuous soil moisture traces (Fig. 66) are interesting for two reasons. Firstly the red and orange traces, showing VSM in the A horizon at depths of 10 and 30 cm respectively indicate permanent saturation for the period June–October 2009 indicating the presence of a shallow perched watertable. The rapid onset of saturation implies water has started flowing through this shallow layer over the dispersive clay layer (Appendix D) in response to the major rainfall events at the end of April 2009. Soils in the region of the access tube remained waterlogged, as wetland stores were filled and rainfall was consistent, meaning water was unable to drain. Saturation persists from early May until the last week of October. By mid November soils have dried considerably, though successive summer rainfall events caused rapid and sustained rise in soil moisture in A horizons.

Also of interest is the lack of change in the 90 and 150 cm soil moisture level. This was the hypothesised expectation for all sites owing to the texture contrast soils, but was only observed in the data for the pasture land cover. This may indicate the influence of sodium in the profile causing dispersion and preventing infiltration. This process would also have acted to seal the access tube, preventing leakage via the outside of the tube to lower parts of the profile. As no saturated layers were present at installation and no change in soil moisture is detected in lower layers, this data suggests that no throughflow occurred through B horizons at the site of the access tube. This was supported by observations during hand auguring for peizometer installation during the project, where no subsoil saturation was observed to depths of over one metre. The confined flow system discussed in Section 5 was the only location of subsoil saturation and was located on the opposite side of the wetland from the access tube.

The surface soil moisture profile for Springs Road (10 cm, red line in Fig. 67) does not maintain persistent saturation. The 30 cm depth appears to have remained saturated for periods of up to two weeks, does not represent the persistent saturation observed at the Wither Swamp. Distinct ranges of variation are observed for wet and dry seasonal periods for both shallow sensors. Topsoil layers increase in moisture consistently from April breaking rains and reach their wettest state early in July. Soils remain in this moisture region until they dry rapidly from around mid October to mid November. Response to rainfall events is evident throughout in the shallow VSM trace. Very low soil moisture content at both 10 cm and 30 cm is evident in the early and late record.

The spike in subsoil moisture levels (90 and 150 cm, green and blue respectively) is thought be most likely be due to down-tube seepage from surface soils once topsoils saturated during major rainfall in July. While throughflow in the B horizon was observed at the site, the rapid response of this vector observed in other studies suggests that some relationship with rainfall would have been anticipated.

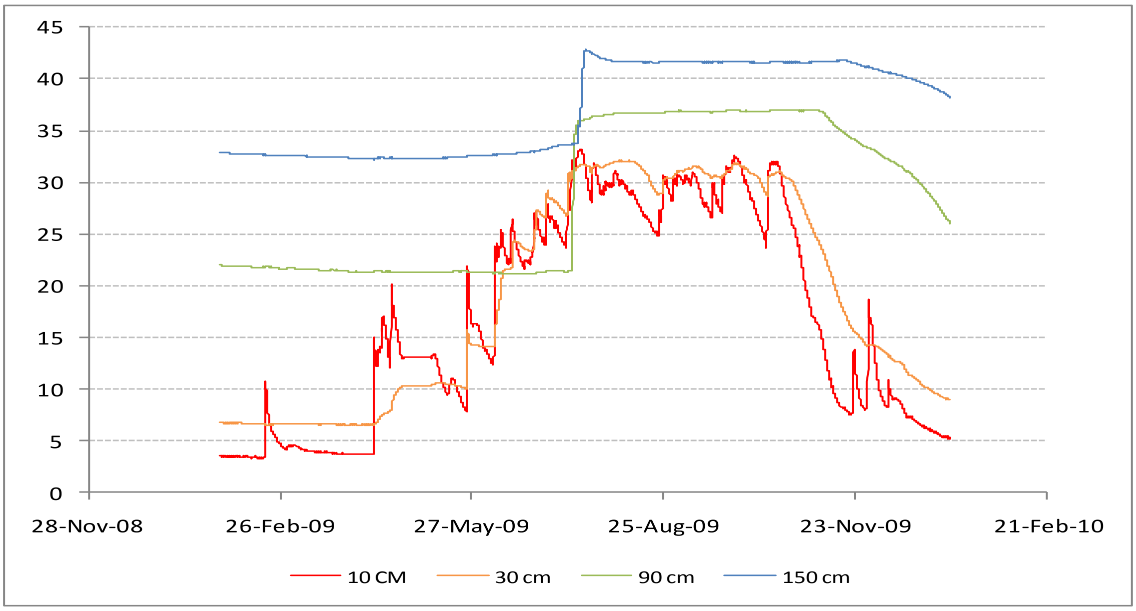


Figure 67. Continuous volumetric soil moisture at four depths – Springs Road

The most dynamic of all the continuous soil moisture data was observed for the Foggy Farm site (Fig. 68). Topsoil layers reach the wet range of variation by early June and remain in this moisture range until mid October when they dry dramatically over the course of about four weeks. As is the case with the Springs Road site it is believed that the spike in 90 and 150 cm soil moisture levels is associated with leakage of water from higher in the profile down the side of the access tube. This is clearly the case in the 90 cm profile which responds to rainfall events concurrently with the 30 cm trace, which would not be possible were seepage via the soil matrix responsible for the rise in moisture. Down-tube seepage has occurred earlier at this site than at Springs Road, possibly due to the disturbance installing the tube, as soft rock was encountered requiring considerable physical force to complete the installation. This increases the potential for air gaps to occur between the tube and the soil profile, later allowing water to move vertically. Hand augering in late spring specifically looking for evidence of B horizon throughflow found no saturated water movement in layers below the surface loams, even adjacent to wetlands.

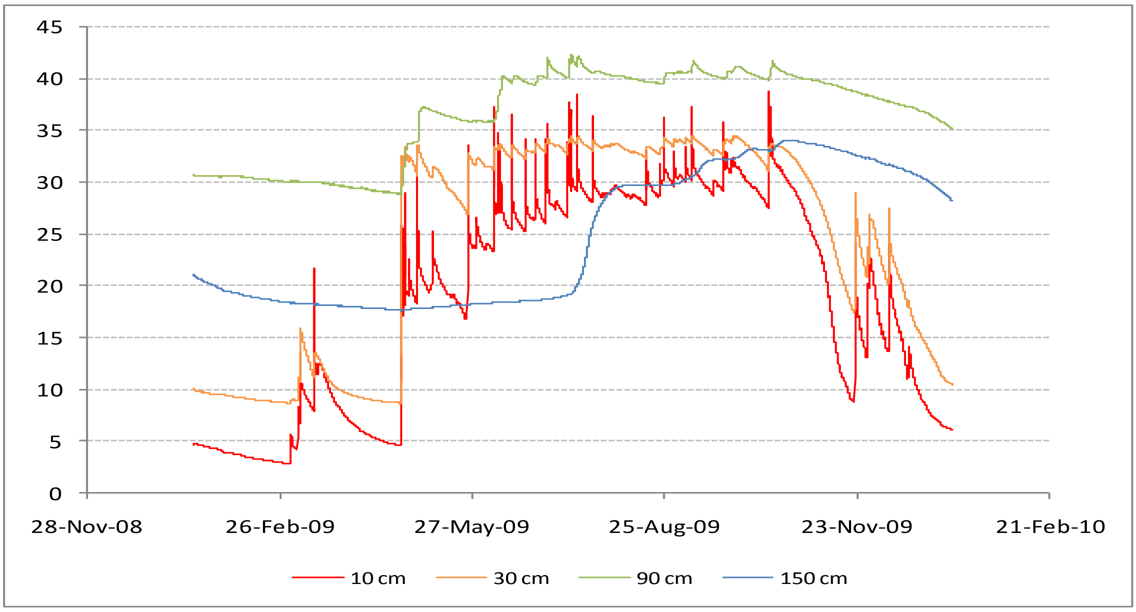


Figure 68. Continuous volumetric soil moisture at four depths – Foggy Farm

J. DESCRIPTION OF WC-1 MODEL

WC-1 is a water balance model developed by David Cresswell (2002) based on experience with South Australian rainfall–runoff calibration in the Mount Lofty Ranges, Barossa Valley and Mid North. The program was developed in 1988 to estimate the impact of farm dams in the Barossa Valley when it was found that most of the existing models tried were not able to reproduce the recorded runoff of South Australia's drier catchments. When annual rainfall lies in the range 450–650 mm, the estimation of runoff becomes a tricky exercise.

Model Concept

WC-1 is a ten-parameter model using three storages as shown in Figure 69 to track interception, soil moisture and groundwater. The soil store is generally the main runoff producing component requiring four parameters for calibration.

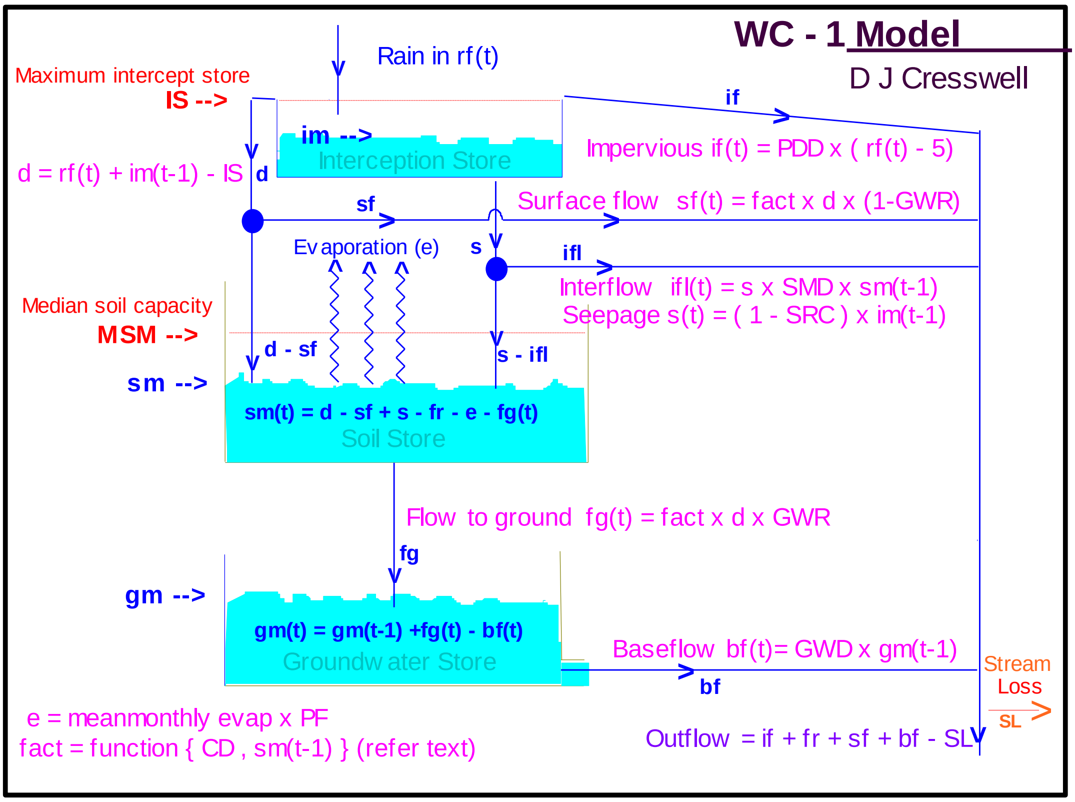


Figure 69. Concept of WC-1 Model

Surface runoff (not including the groundwater contribution) is calculated with both a Hortonian and saturated surface area component. The Hortonian component is generally small and is calculated as the runoff from an impervious area that has a daily loss rate of 5 mm. The parameter PDD is used to input the fraction of the catchment contributing to runoff.

By far the greatest proportion of surface flow is generated by calculating the saturated surface area of the catchment. To do this, the model tracks the soil storage and calculates the area saturated based on the assumption that the soil moisture-holding capacity is normally distributed across the catchment. This is shown in Figure 70.

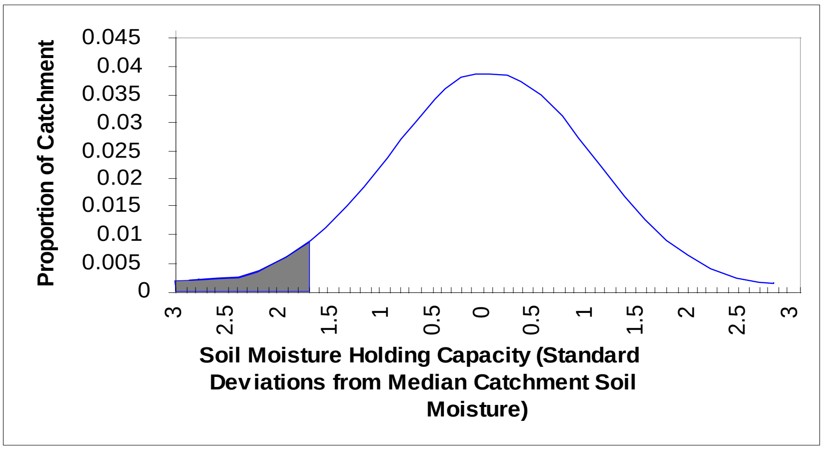


Figure 70. Proportion of catchment contributing to runoff, calculated from soil moisture

To calibrate such a model two parameters are required—the median soil moisture of the catchment (MSM) and the catchment standard distribution (CD). Typically, these values are found to lie in the range 150–250 mm (MSM) and 20–80 mm (CD).

When dry, the soil moisture lies >3 standard deviations to the left of the median centre and as the catchment wets up this moves towards the fully saturated catchment, which occurs at median soil moisture plus 3 standard deviations. At any point on the axis, the proportion of catchment assumed to be saturated is calculated as the area under the normal distribution curve.

For example, Figure 70 indicates that when the soil moisture of the soil store reaches MSM-1.6xCD the area shaded is the proportion of the catchment contributing to the runoff. From normal distribution tables this is 5.5% of the catchment.

When the median soil moisture is reached, the proportion of catchment contributing to runoff is 50%, as shown in Figure 71.

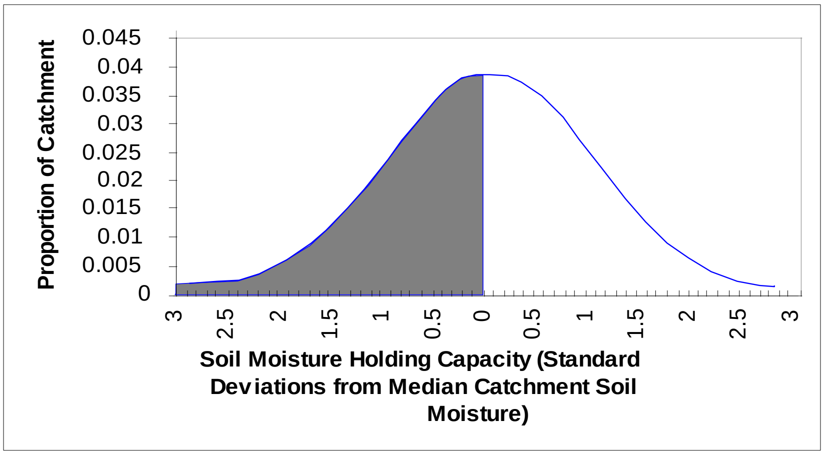


Figure 71. Proportion of catchment contributing to runoff calculated from soil moisture

The shape of this relationship (Fig. 72 is similar to a power curve but asymptotic to Y = 0 and Y = 1. Intuitively this is what is expected and overcomes the problem of the power curve that is required to be silled at 1.0.

The volume of water running off the catchment is then the product of the contributing area and the effective rainfall. Catchments in semi-arid areas show a capacity to retain quite significant rainfall events, requiring the use of an interception store for accurate simulation.

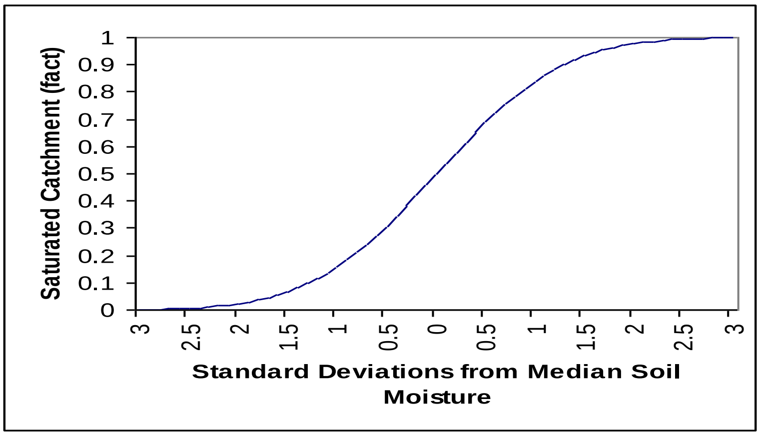


Figure 72. Proportion of catchment contributing to runoff calculated from soil moisture

The effective rainfall is defined as the volume of water spilling the interception store.

The maximum interception store (IS) may typically range from zero to 30 mm and is tracked continuously within the model. Water may leave the interception storage either by overtopping the storage, thus becoming effective rainfall, or it may percolate slowly into the soil store where it contributes to an interflow component of flow. This percolation occurs at a rate calculated in a similar way to the Annual Precipitation Index (API).

The transfer rate is independent of season and is set by the soil wetness multiplier (SWM), typically to a value of 0.9. The value set is the proportion of the water held in the store (im(t)) which is retained to the next day. Seepage is calculated as:

S = (1-SWM) x im(t)

During the wet season, the baseflows of the streams are seen to rise but the duration of such flows remains dependent on relatively continuous rain falling on the catchment. It is proposed that this baseflow return occurs due to the over-saturated areas of the catchment returning a fraction of this moisture back to the streams. As the catchment dries or during long spells of no rain, it is expected that this return will drop to zero.

This interflow is assumed in the model to be:

Ifl = s x SMD x sm(t)

SMD is the parameter defining the proportion returned to the stream.

The catchment response is therefore defined by the six parameters mentioned above but evaporation can potentially override all of these. In semi-arid catchments, choosing the correct evaporation rate is critical.

Models use various formulas ranging from linear to power functions to estimate the moisture loss from soils. Experimentation with the linear model was not found to improve the estimate of runoff and was discarded for the simpler constant model. Here, evapotranspiration is assumed to equal the pan factor multiplied by recorded daily evaporation. Typically a value of 0.6 to 0.7 is used for class A pan recordings.

Groundwater is simulated within the model using two parameters — GWR (recharge) and GWD (discharge). Both operate in a simple linear fashion.

Groundwater recharge is seen to have a greater relationship with streamflow than total rainfall. This suggests that groundwater recharge requires similar conditions to streamflow, hence the wetting up of the catchment, to occur. Tying recharge to streamflow simulates this, thereby assuming that the greater the saturated catchment-generated streamflow occurring, the more recharge occurs from the soil to the groundwater store.

The parameter GWR (Groundwater Recharge) is used to define the proportion passing to ground and this may often be up to 20–30%.

Baseflow discharging from the groundwater store is simply a linear relationship defined by parameter GWD (Groundwater Discharge). No loss is assumed to occur from the groundwater store to external basins.

Summary of WC-1 Parameters

Medium soil moisture (MSM) — Represents the field capacity of the soil. Usually in the range of 150–300 mm. Increasing this value delays the early season initiation of runoff, decreases runoff by providing greater opportunity for evapotranspiration and assists in keeping late season groundwater flows up.

Interception store (IS) — Represents the maximum initial abstraction from rainfall before any runoff can occur. The normal range is 10–25 mm. A larger value will inhibit runoff after dry spells and reduce the total amount of runoff.

Catchment distribution (CD) — Sets the range of soil moisture values about MSM. Usual values are 25–60 mm. A larger value will initiate runoff earlier and more often.

Groundwater discharge (GWD) — proportion of the groundwater store that discharges as baseflow to the stream. This is a simple linear function:

Baseflow = groundwater store x GWD

Usual values are small (0.001 to 0.0001).

Soil moisture discharge (SMD) — As soil moisture increases there is a rise in the baseflow that occurs due to the saturation of the soil storage. Values are usually small (0.0001).

Pan factor for soil (PF) — factor applied to the daily evaporation calculated from the monthly pan evaporation data. The usual range is 0.6–1.0. The higher the value, the less the runoff. The higher the value, the earlier runoff ceases after winter.

Proportion direct drainage (PDD) — proportion of the catchment that can be considered relatively impervious. After an initial loss of 5 mm, rainfall on this area will be discharged as surface flow. Usual values for this are zero.

Store wetness multiplier (SWM) — value determines the rate that water from the interception store moves to the soil store. The transfer rate is independent of season and ensures that the amount of water retained in the interception store follows a similar power recession curve of the Annual Precipitation Index (API). Usual values are ~0.9.

Groundwater recharge (GWR) — proportion of rainfall that recharges the groundwater store. Usual values are 0.05–0.3, indicating that 5–30% of the flow running off the catchment is entering the groundwater system.

Creek loss (CL) — a reduction factor used to decrease runoff. It is generally set to zero.

UNITS OF MEASUREMENT

Units of measurement commonly used (SI and non-SI Australian legal)

Name of unit	Symbol	Definition 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
decimetre	dm	10^{-1} m	length
microgram	µg	10^{-6} g	mass
microlitre	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

Shortened forms

~	approximately equal to	ppb	parts per billion
bgs	below ground surface	ppm	parts per million
EC	electrical conductivity (µS/cm)	ppt	parts per trillion
K	hydraulic conductivity (m/d)	w/v	weight in volume
pH	acidity	w/w	weight in weight

GLOSSARY

Aquatic ecosystem — The stream channel, lake or estuary bed, water and/or biotic communities and the habitat features that occur therein

Aquatic habitat — Environments characterised by the presence of standing or flowing water

Aquatic macrophytes — Any non-microscopic plant that requires the presence of water to grow and reproduce

Aquifer — An underground layer of rock or sediment that holds water and allows water to percolate through

Artesian — An aquifer in which the water surface is bounded by an impervious rock formation; the water surface is at greater than atmospheric pressure and hence rises in any well which penetrates the overlying confining aquifer

Baseflow — The water in a stream that results from groundwater discharge to the stream; often maintains flows during seasonal dry periods and has important ecological functions

Biodiversity — (1) The number and variety of organisms found within a specified geographic region. (2) The variability among living organisms on the earth, including the variability within and between species and within and between ecosystems

Biological diversity — See 'biodiversity'

Biological integrity — Functionally defined as the condition of the aquatic community that inhabits unimpaired water bodies of a specified habitat as measured by community structure and function

Biota — All of the organisms at a particular locality

BoM — Bureau of Meteorology, Australia

Bore — See 'well'

Buffer zone — A neutral area that separates and minimises interactions between zones whose management objectives are significantly different or in conflict (eg. a vegetated riparian zone can act as a buffer to protect the water quality and streams from adjacent land uses)

Catchment — That area of land determined by topographic features within which rainfall will contribute to run-off at a particular point

Confining layer — A rock unit impervious to water, which forms the upper bound of a confined aquifer; a body of impermeable material adjacent to an aquifer; see also 'aquifer, confined'

Critical habitat — Those areas designated as critical for the survival and recovery of threatened or endangered species

CSIRO — Commonwealth Scientific and Industrial Research Organisation

Dams, on-stream dam — A dam, wall or other structure placed or constructed on, in or across a watercourse or drainage path for the purpose of holding and storing the natural flow of that watercourse or the surface water

Diversity — The distribution and abundance of different kinds of plant and animal species and communities in a specified area

d/s — Downstream

DWLBC — Department of Water, Land and Biodiversity Conservation (Government of South Australia)

EC — Electrical conductivity; 1 EC unit = 1 micro-Siemen per centimetre (µS/cm) measured at 25°C; commonly used as a measure of water salinity as it is quicker and easier than measurement by TDS

Ecological indicators — Plant or animal species, communities, or special habitats with a narrow range of ecological tolerance; for example, in forest areas, such indicators may be selected for emphasis and monitored during forest plan implementation because their presence and abundance serve as a barometer of ecological conditions within a management unit

Ecological processes — All biological, physical or chemical processes that maintain an ecosystem

Ecological values — The habitats, natural ecological processes and biodiversity of ecosystems

Ecology — The study of the relationships between living organisms and their environment

Ecosystem — Any system in which there is an interdependence upon and interaction between, living organisms and their immediate physical, chemical and biological environment

Endemic — A plant or animal restricted to a certain locality or region

Environmental water provisions — That part of environmental water requirements that can be met; what can be provided at a particular time after consideration of existing users' rights and social and economic impacts

Environmental water requirements — The water regimes needed to sustain the ecological values of aquatic ecosystems, including their processes and biological diversity, at a low level of risk

Ephemeral streams or wetlands — Those streams or wetlands that usually contain water only on an occasional basis after rainfall events. Many arid zone streams and wetlands are ephemeral.

Evapotranspiration — The total loss of water as a result of transpiration from plants and evaporation from land and surface water bodies

Flow bands — Flows of different frequency, volume and duration

Flow regime — The character of the timing and amount of flow in a stream

Fresh — A short duration, small volume pulse of streamflow generated by a rainfall event that temporarily, but noticeably, increases stream discharge above ambient levels

Geomorphic — Related to the physical properties of the rock, soil and water in and around a stream

Geomorphology — The scientific study of the landforms on the Earth's surface and of the processes that have fashioned them

GIS — Geographic Information System; computer software linking geographic data (for example land parcels) to textual data (soil type, land value, ownership). It allows for a range of features, from simple map production to complex data analysis

Gley, gleyed – a form of hydric soil with blue-grey coloration caused by waterlogging. Characteristic of wetland soils. Will become mottled grey and yellow when exposed to air.

Groundwater — Water occurring naturally below ground level or water pumped, diverted and released into a well for storage underground; see also 'underground water'

Habitat — The natural place or type of site in which an animal or plant, or communities of plants and animals, live

Throughflow - the movement of water horizontally through the soil or regolith. The term implies water travels via surface flow processes to surface water bodies, which distinguishes it from interflow

Hydrogeology — The study of groundwater, which includes its occurrence, recharge and discharge processes and the properties of aquifers; see also 'hydrology'

Hydrology — The study of the characteristics, occurrence, movement and utilisation of water on and below the Earth's surface and within its atmosphere; see also 'hydrogeology'

Hydrometric — Literally relating to water measurement, from the Greek words 'hydro' (water) and metrikos (measurement)

Hydro-period – The duration of inundation of a wetland, typically over an annual cycle

Impact — A change in the chemical, physical, or biological quality or condition of a water body caused by external sources

m AHD — Defines elevation in metres (m) according to the Australian Height Datum (AHD)

Metadata — Information that describes the content, quality, condition and other characteristics of data, maintained by the Federal Geographic Data Committee

MLR — Mount Lofty Ranges

Model — A conceptual or mathematical means of understanding elements of the real world that allows for predictions of outcomes given certain conditions. Examples include estimating storm run-off, assessing the impacts of dams or predicting ecological response to environmental change

Monitoring — (1) The repeated measurement of parameters to assess the current status and changes over time of the parameters measured (2) Periodic or continuous surveillance or testing to determine the level of compliance with statutory requirements and/or pollutant levels in various media or in humans, animals and other living things

Native species — Any animal and plant species originally in Australia; see also 'indigenous species'

Natural recharge — The infiltration of water into an aquifer from the surface (rainfall, streamflow, irrigation etc). See also recharge area, artificial recharge

Natural resources — Soil, water resources, geological features and landscapes, native vegetation, native animals and other native organisms, ecosystems

NRM — Natural Resources Management; all activities that involve the use or development of natural resources and/or that impact on the state and condition of natural resources, whether positively or negatively

Observation well — A narrow well or piezometer whose sole function is to permit water level measurements

Pasture — Grassland used for the production of grazing animals such as sheep and cattle

Percentile — A way of describing sets of data by ranking the dataset and establishing the value for each percentage of the total number of data records. The 90th percentile of the distribution is the value such that 90% of the observations fall at or below it.

Perennial streams — Permanently inundated surface stream courses. Surface water flows throughout the year except in years of infrequent drought.

Permeability — A measure of the ease with which water flows through an aquifer or aquitard, measured in m^2/d

Phreatophytic vegetation — Vegetation that exists in a climate more arid than its normal range by virtue of its access to groundwater

Piezometer — A narrow tube, pipe or well; used for measuring moisture in soil, water levels in an aquifer, or pressure head in a tank, pipeline, etc

Pluviometer — An automated rain gauge consisting of an instrument to measure the quantity of precipitation over a set period of time

Potentiometric head — The potentiometric head or surface is the level to which water rises in a well due to water pressure in the aquifer, measured in metres (m); also known as piezometric surface

Prescribed water resource — A water resource declared by the Governor to be prescribed under the Act and includes underground water to which access is obtained by prescribed wells. Prescription of a water resource requires that future management of the resource be regulated via a licensing system.

Recharge area — The area of land from which water from the surface (rainfall, streamflow, irrigation, etc.) infiltrates into an aquifer. See also artificial recharge, natural recharge

Riparian — Of, pertaining to, or situated or dwelling on the bank of a river or other water body

Seasonal watercourses or wetlands — Those watercourses or wetlands that contain water on a seasonal basis, usually over the winter–spring period, although there may be some flow or standing water at other times

Sub-catchment — The area of land determined by topographical features within which rainfall will contribute to run-off at a particular point

Subsoil — the soil profile layer between the topsoil and the substrate equivalent to a B horizon (Hall *et al.* 2009)

Surface water — (a) water flowing over land (except in a watercourse), (i) after having fallen as rain or hail or having precipitated in any another manner, (ii) or after rising to the surface naturally from underground; (b) water of the kind referred to in paragraph (a) that has been collected in a dam or reservoir

Sustainability — The ability of an ecosystem to maintain ecological processes and functions, biological diversity and productivity over time

Taxa — General term for a group identified by taxonomy, which is the science of describing, naming and classifying organisms

TDS — Total dissolved solids, measured in milligrams per litre (mg/L); a measure of water salinity

Throughflow – the horizontal movement of water in the soil zone.

Underground water (groundwater) — Water occurring naturally below ground level or water pumped, diverted or released into a well for storage underground

u/s — Upstream

WAP — Water Allocation Plan; a plan prepared by a CWMB or water resources planning committee and adopted by the Minister in accordance with the Act

Watercourse — A river, creek or other natural watercourse (whether modified or not) and includes: a dam or reservoir that collects water flowing in a watercourse; a lake through which water flows; a channel (but not a channel declared by regulation to be excluded from the this definition) into which the water of a watercourse has been diverted; and part of a watercourse

Water-dependent ecosystems — Those parts of the environment, the species composition and natural ecological processes, that are determined by the permanent or temporary presence of flowing or standing water, above or below ground; the in-stream areas of rivers, riparian vegetation, springs, wetlands, floodplains, estuaries and lakes are all water-dependent ecosystems

WDE — Water dependent ecosystem

Well — (1) An opening in the ground excavated for the purpose of obtaining access to underground water. (2) An opening in the ground excavated for some other purpose but that gives access to underground water. (3) A natural opening in the ground that gives access to underground water

Wetlands — Defined by the Act as a swamp or marsh and includes any land that is seasonally inundated with water. This definition encompasses a number of concepts that are more specifically described in the definition used in the Ramsar Convention on Wetlands of International Importance. This describes wetlands as areas of permanent or periodic to intermittent inundation, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tides does not exceed six metres.

WMLR — Western Mount Lofty Ranges

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